The Effects of Interim Operational Plan on the Mainland Mangrove Wetlands of Florida Bay: Hydrology, Hydrography, Fish Community Dynamics and Roseate Spoonbill

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1. Introduction

Implementation of Test Iteration 7 of the Experimental Program of Water Deliveries to Everglades National Park (ENP) began October 1, 1995 (USACOE 1995). The Environmental Assessment (EA) for Test 7 spelled out the design and terms of the test iteration. The EA specified that an integral part of the test was the establishment of a monitoring program to assess ecological responses during the period of Test 7. Four criteria were used to rate various potential monitoring projects (USACOE 1995). Primary among the criteria was the potential sensitivity of the monitoring protocol to hydrological change. The other three criteria were; 1, the existence of baseline data so that evaluations could be made 2, relevance to ecological modeling efforts (specifically the ATLSS modeling effort) and 3, significance to Section 7 of the Endangered Species Act. Based on these four criteria, Audubon of Florida's Mangrove Ecosystem Research Project (MERP) was rated as essential to the Test 7 monitoring program (USACOE 1995) and was subsequently contracted by the U.S. Army Corp of Engineers (USACOE) and ENP. Under the auspices of the Test 7 monitoring program, the charge to the MERP was to evaluate the relative impact of Test 7 on the fishes of the mangrove transition zone between the fresh water environs of the Taylor Slough/C-111 basins and Florida Bay.

In November 1999, the USACOE terminated the Experimental Program of Water Deliveries to ENP and adopted the Interim Structural and Operational Plan (ISOP) as an emergency measure designed to protect the Cape Sable Seaside Sparrow (CSSS) while still providing adequate water supplies the Park. In January 2001, changes were made to the original ISOP to enhance ecological function within ENP. This new plan was referred to as ISOP 2001 (to differentiate from the original ISOP 2000). ISOP 2001 was continued through June of 2002. In June 2002, The USACOE issued the Final Environmental Impact Statement that began operations under Interim Operational Plan (IOP). The MERP was continued under ISOP 2000, ISOP 2001 and IOP with the same objectives as under Test 7.

Analyses were conducted on MERP data as they were being collected (Lorenz 2000). These analyses indicated two major effects of upstream water management practices on the mangrove ecosystem bordering the northern shoreline of eastern Florida Bay. The first was that diversion of natural flows resulted in alterations of the salinity regime in the mangrove zone (McIvor et al. 1994; Lorenz 2000, Appendix 3). The change in salinity patterns negatively affected primary production in the submerged aquatic vegetation community within the mangrove zone (Montague and Ley 1993; Frezza and Lorenz 2003), which, in turn, resulted in lower abundance of prey base fishes (Ley et al. 1994; Lorenz 1997; Lorenz 2000). The observed decline in productivity within the prey base fishes likely explains declines in predator populations dependant on this food resource (Bancroft et al. 1994; Lorenz et al. 2002). Supporting evidence for the relationship between water management, salinity, and ecological productivity can be found in Appendix 1.

The second major effect of water management was that pulse releases of water from the canal system during the dry season resulted in reversals of the seasonal drying patterns within the coastal mangrove ecosystem. Drying events are critical to the ecosystem in that myriad predators take advantage of a highly abundant food source in the form of prey base fishes concentrated into the remaining small pools (Kushlan 1976a; Kushlan 1976b; Loftus and Kushlan 1987; Lorenz 2000). Wading birds (including Roseate Spoonbills) gear the timing of nesting to these drying events which enables them to readily meet the high energetic requirements of their rapidly growing young (Frederick and Colopy 1989; Bjork and Powell 1994; Lorenz 2000; Dumas 2000). Anthropogenic reversals in the drying pattern allow prey base fishes to spread out across the landscape making them relatively unavailable to wading birds, thereby resulting in nesting failure (Lorenz 2000). The depth at which fish begin to move into deeper habitats (i.e., begin to concentrate) is when water levels on the wetland surface drops below 12.5 cm or 5 inches (Lorenz 2000, Appendix 1).

Based on these previous findings, desirable conditions that promote a healthy ecosystem can be identified. Ideally, there should be a great deal of intra-annual variation in hydrology and hydrography. During the wet season, lower salinity within the mangrove wetlands is desirable. High wet season water levels and relatively large flows through Taylor Slough would result in the desired condition (Appendix 2). Prey fishes would be expected to respond by increasing numbers under these low salinity conditions. During the dry season, low water levels and curtailment of reversals are desirable. These conditions would be promoted by reduced flows through the system especially through the C-111 canal (Appendix 2). The expected biological response to these conditions would be relatively high availability of prey and relatively high reproductive output by spoonbills.

This report will examine the effect of ISOP/IOP relative to Test 7 on both of these major ecosystem relationships using the Before-After Control-Impact (BACI) analytical technique. Supporting evidence for the relationship between water management, water depth, availability of fish and nesting success can be found in Appendix 1.

2. Methods Pertinent to BACI Analyses

In April 2003, environmental scientists working within ENP met to discuss the approach to be used in evaluating IOP. This group set up the basic definitions and parameters to be used. A consensus was reached that the best approach to evaluating the impact of ISOP/IOP was to use the BACI method and that data collected by the MERP were ideally suited for this analytical technique.

It was decided that the “Before” period would be Test 7 (November 1995 to Oct 2000) and the “After” period would be ISOP/IOP (November 1999 to October 2001). The wet season was defined as June 1 to October 31 and the dry season was from November 1 to May 31. The hydrologic year was defined from the beginning of the wet season (June 1) to the end of the dry season (May 31). Unfortunately, results from the MERP did not conform exactly to these definitions so minor deviations were made such that ecological meaningful analyses would result. These changes were necessary because only whole hydrologic years are meaningful with the fish/spoonbill data (i.e., wet season conditions dictate dry season responses). Therefore, the hydrologic year 1999-2000 was omitted from the data set since the wet season fell under Test 7 while the dry season fell under ISOP. Furthermore, the dry season of 1995-1996 was dropped because the prior wet season was not within the "Before" period (it fell under Test 6). This resulted in three complete years in the "Before" period but only 2 in the "After." This created problems in sample size so the dry season of 2002-2003 was appended resulting in 3 complete years in both the "Before' and "After" periods. Thus; for the analysis of MERP data, the "Before" period was defined as the three hydrologic years from June 1996 to May 1999 and the "After" period was defined as the three hydrologic years from June 2000 to May 2003.

The location of the fish sampling sites is presented in Figure 1. Three "Impact" sampling sites were located just north of Florida Bay in the Taylor Slough/C-111 drainage area of ENP (Figure 1). As such these sites were heavily impacted by water management operations. These sites were located north of Little Madeira Bay on Taylor River (TR: 25° 13.20'N by 80° 39.00'W), north of the eastern end of Joe Bay (JB: 25° 15.00'N by 80° 31.92'W), and on a tributary west of Highway Creek (HC: 25° 15.25'N by 80° 27.28'W) near US Highway 1.

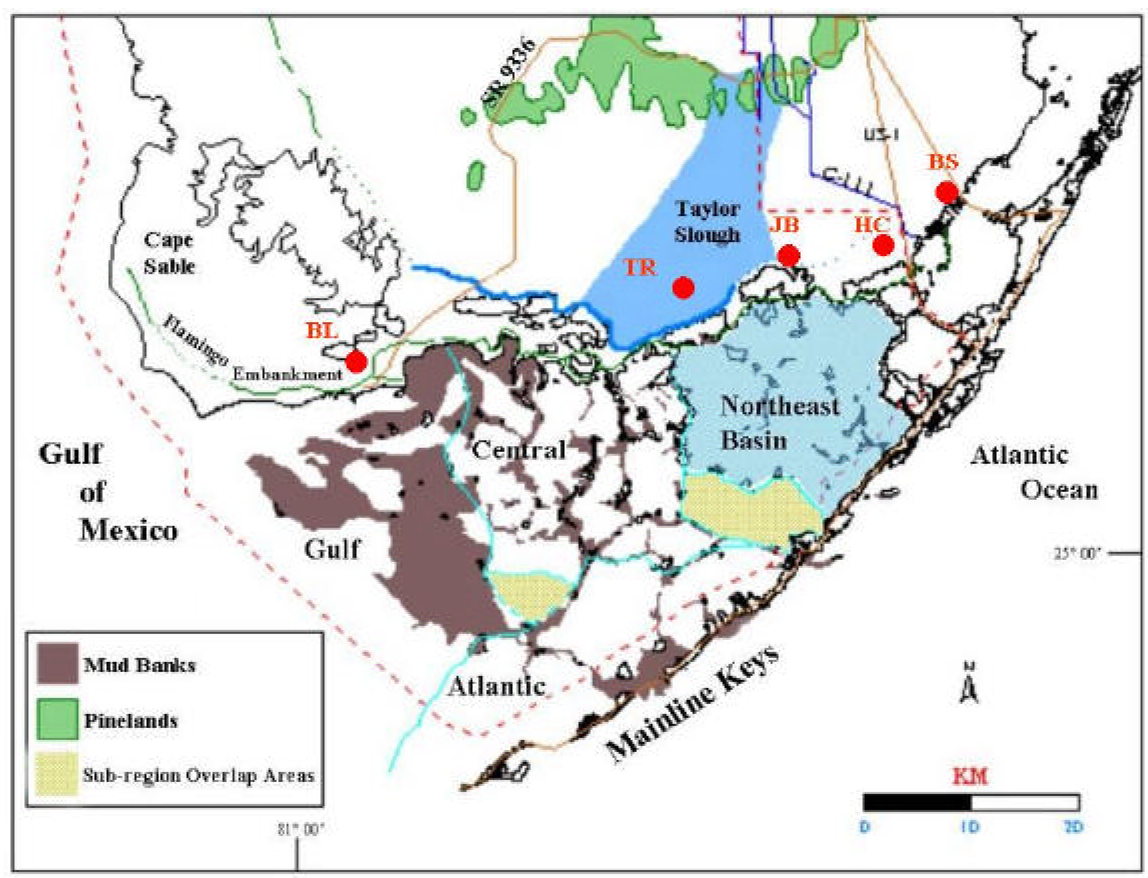

Figure 1. Site locations for hydrostations and fish collections.

The eastern most fish-sampling site was located outside of ENP on Biscayne Bay near the western end of Card Sound Bridge on Barnes Sound (BS: 25° 17.65N by 80° 24.57'W). This site was considered the control site for salinity analyses because it was impounded by roadbeds (US-1 and Card Sound Road) and received no freshwater sheet flow from the Everglades. As a result, all freshwater input to BS was from local rainfall or groundwater seepage, i.e., water management operational changes have only minimal impacts on the hydrography of this site. Unfortunately, this site is influenced by diurnal tides while the "Impact" sites were non-tidal. This confounds the analyses of water depth so this site could not be considered a control for water depth.

A fifth site was located on Cape Sable on the southeastern shore of Bear Lake (Figure 1.; BL: 25°09.68'N by 80°57.11'W). This site was hydrologically isolated from both tidal inundation from the marine environment and from direct water management impacts from either Test 7 or ISOP/IOP. This isolation makes this site an ideal control site for water level analyses. However, since the site is so remote from the marine environment, the salinity regime is very different from the "Impact" sites regardless of water management practices. Therefore, the site cannot be used as a salinity "control."

Rainfall data, acquired from Hosung Ahn at ENP, were the agreed upon standard. Discharge from the C-111 was calculated as flow through Structure S-18C minus flow through Structure S-197. These data were acquired from Kevin Kotun at ENP. Taylor Slough flow was monitored directly at Taylor Slough Bridge by ENP staff and was also acquired from Kevin Kotun. Water level and salinity data were collected at each of the five sites and were analyzed using rainfall as a control. The problem of mixed units (inches for rainfall, PSU for salinity and cm for depth) was solved by relativizing the data sets on a zero-to-one scale (McCune and Grace 2002). BACI analyses were performed on the relativized data, however, for clarity of presentation, the un-relativized raw data are presented in the figures of analytical results, i.e., figures show actual measured units but BACI results presented with the figures were performed on relativized data.

Fish collections were made at all five sites in June, September and monthly from November to April. Thus, six fish samples were collected at each site in each dry season while only two were collected during wet seasons. Fish abundance and availability were estimated from collections using a 9m^2 drop trap (Lorenz et al. 1997, Appendix 3). Fish abundance is defined as the stratified mean across the entire landscape (i.e., it discounts the concentration effect). Fish availability is defined as the mean number of fish collected in the strata with the highest abundance (i.e., prey availability is a measure of the prey concentration effect caused by the drawing down of wetland surface water). Fish abundance and availability estimates were square root transformed so as to meet the assumption of normality required for BACI Analysis of Variance (McCune and Grace 2002). However for clarity of presentation, the un-transformed raw data are presented in the figures of analytical results, i.e., figures show actual measured estimates of fish density but BACI results presented with the figures were performed on square-root transformed data.

Spoonbill nesting success data was collected at two sites (Tern and Sandy keys) as per methods in Appendix 3. Tern Key is located directly downstream of the Taylor Slough/ Panhandle region. These birds feed in the coastal mangroves represented by the "Impact" sites and is considered the impacted colony. Sandy Key is located on the western edge of Florida Bay. These birds feed in the Cape Sable area (as represented by the BL site) and are considered the control site since this is not directly impacted by water management practices.

To summarize the BACI analysis: Analysis Of Deviance (ANODE) were performed on rainfall, flow, salinity, water level, fish abundance, fish availability. Spoonbill nesting success was analyzed using Analysis of Variance (ANOVA). The three Test 7 hydrologic years were considered the "Before" period while the three ISOP/IOP years were considered the "After" period. Rainfall and data collected at BS were used as the salinity control and rainfall and data collected at BL sites were considered the control for water depth. Salinity and water level data collected at TR, JB, and HC sites were used as "Impact." ANODE values of greater than 2 were considered significant. Significant results were followed by *post apriori* tests using the Least Squared Means technique (Statistica 1999). The critical range for rejecting null hypotheses using ANOVA or *post apriori* tests was set at $p<0.10$.

2.1 Rainfall and Flow

A comparison of rainfall between the "Before" (Test 7) and "After" (IOP) periods indicated that there was no significant difference between the periods. Figure 2 indicates that the three Test 7 years compared well with the three IOP years. Both 1996-97 and 2000-01 were relatively low rainfall years while 1997-98 and 2001-02 were relatively high rainfall years. Both 1998-99 and 2002-03 were moderate rainfall years, however, 1998-99 was appreciably (although not statistically) lower than 2002-03. The temporal distribution of rain through the year was also not significantly different between the two periods (Figure 3). Given that the "Before" and "After" periods had similar rainfall patterns, any differences observed between the periods in other parameters may be attributable to the different water management practices between periods, i.e., Test 7 and ISOP/IOP.

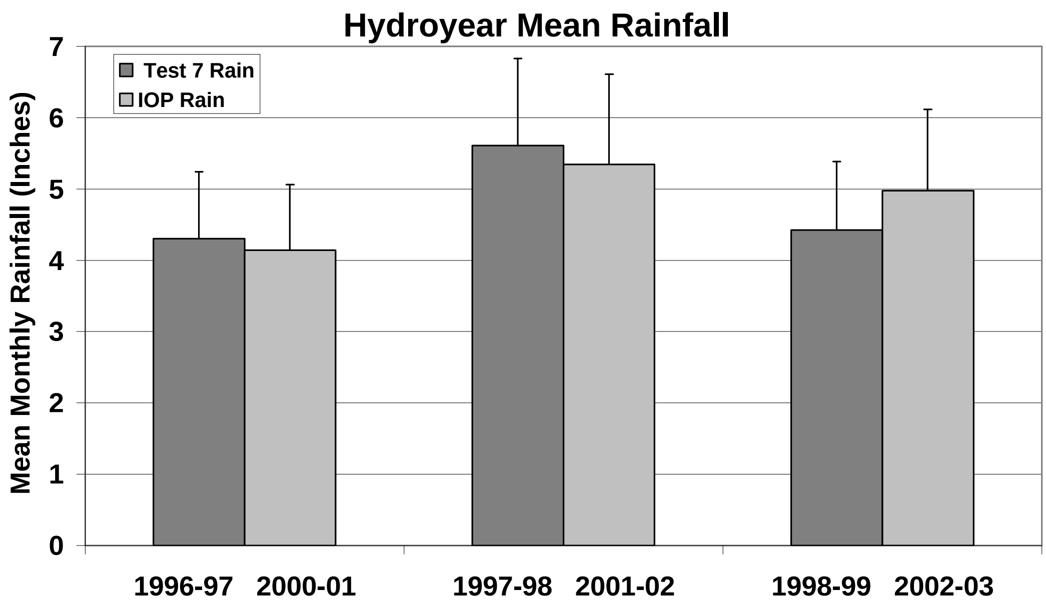

Figure 2. Mean monthly rainfall (±SE) for each hydrologic year used in the BACI analyses. ANOVA indicates no significant differences between years and operational plans ($F_{(5,66)}$=0.31, p=0.9074).

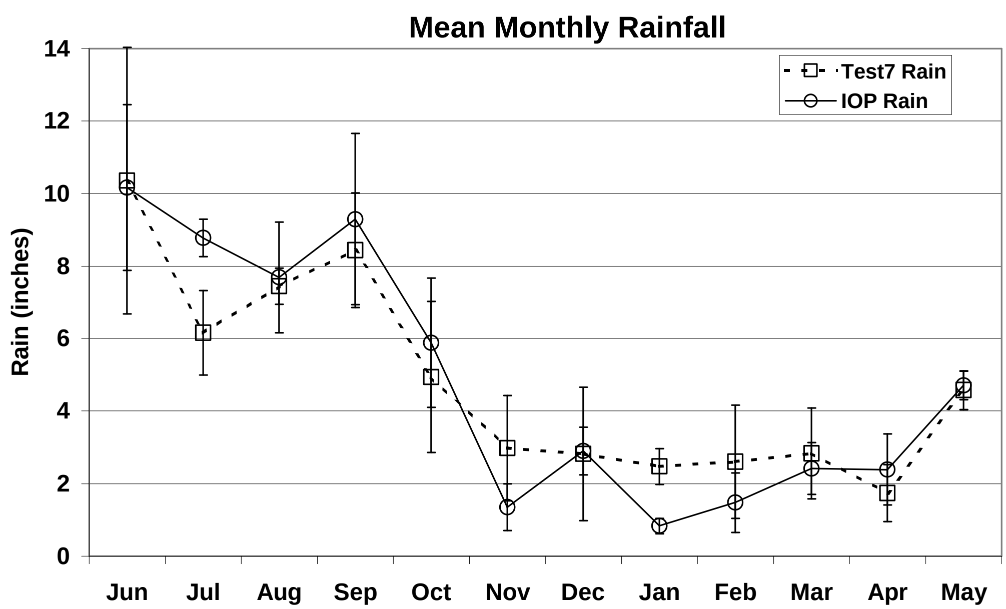

Figure 3. Mean rainfall (+se) for each month by operational plan. Two-Way ANOVA indicated no significant interaction between months and operational plan ($F_{(11,48)}$=0.33; p<0.9754). Least Squares Means indicated that there was no difference between operational plans for each individual month.

There were no significant differences in flow between the Test 7 and IOP periods for either C-111 discharge or Taylor Slough flow (Figure 4). However, during the wet season, there was significantly higher flow to C-111 during IOP compared to Test 7 (Figure 5). This suggests that Test 7 was better than IOP at reducing discharge at C-111. This is a desirable effect based on analyses of mangrove salinity patterns in response to basin flows. Previous analyses (Appendix 2) indicate that wet season flows through Taylor Slough reduces salinity in the mangrove zone while wet season flows through C-111 have minimal salinity impact. Given that reduced salinity is a desirable effect, these data suggest that Test 7 was more beneficial to the mangrove ecosystem than the IOP.

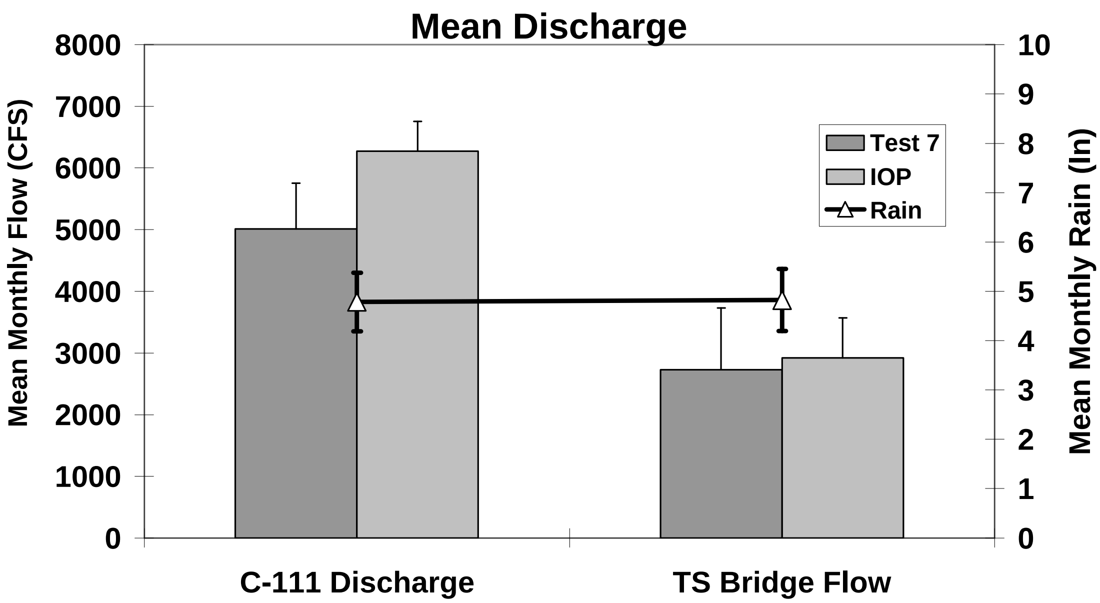

Figure 4. Mean monthly flow (+se) at Taylor Slough Bridge and discharge at the C-111 canal for each operational period. Mean monthly rainfall (+se) is also indicated. Two-Way ANOVA for operational plan and flow basin (including rainfall) were not significant $F_{(2,203)}=0.66$; $p<0.5203$. Least squares means indicated no differences between operational plans for either basin or rainfall.

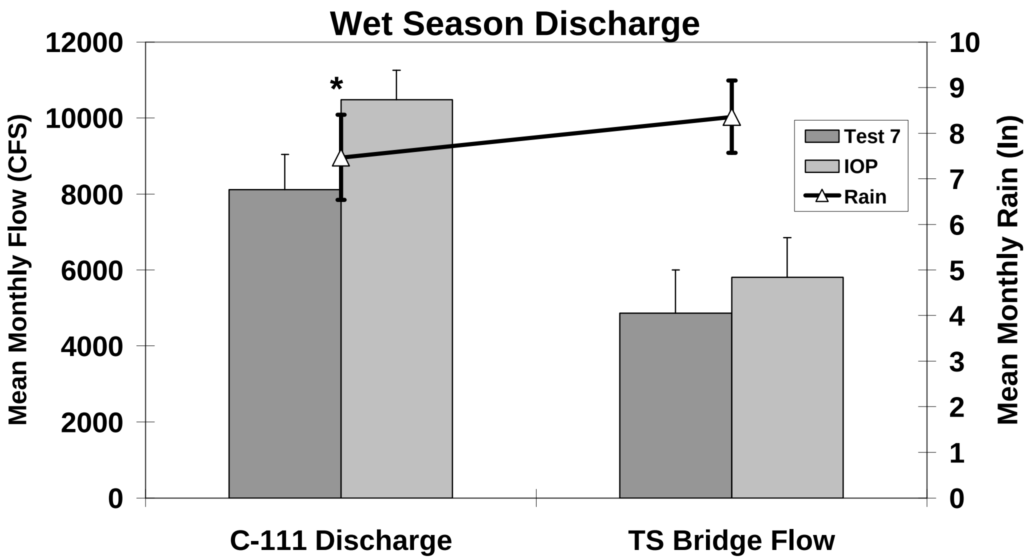

Figure 5. Wet season mean monthly flow (+se) at Taylor Slough Bridge and discharge at the C-111 canal for each operational period. Mean monthly rainfall (+se) is also indicated. Two-Way ANOVA for operational plan and flow basin (including rainfall) were not significant $F_{(2,83)}=1.11$; $p<0.3339$. Least squares means indicated no differences between operational plans for Taylor Slough or rainfall, however, there was a significant difference in wet season discharge at C-111 (*$p<0.05$)

During the dry season, no statically significant differences were observed (Figure 6), between IOP and Test 7, however, the higher discharge rate at C-111 under Test 7 may be biologically relevant in that dry-season discharges may cause reversals in the dry-down patterns on the coastal wetlands, thereby disrupting foraging patterns in wading birds (Lorenz 2000; Gawlik 2002).

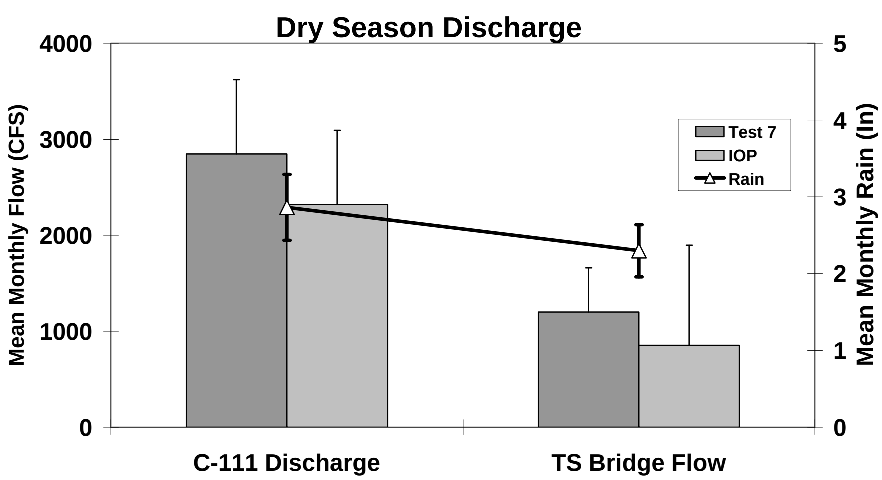

Figure 6. Dry season mean monthly flow (+se) at Taylor Slough Bridge and discharge at the C-111 canal for each operational period. Mean monthly rainfall (+se) is also indicated. Two-Way ANOVA for operational plan and flow basin (including rainfall) were not significant $F_{(2,114)}=0.16$; $p<0.8559$. Least squares means indicated no differences between operational plans for either basin or rainfall during the dry season.

3. Salinity Effects: Results and Discussion:

To test the efficacy of using the BS site as a salinity control, salinity at BS was compared to rainfall (Figure 7). Not only was there no statistical difference between the two periods but the mean and variance in salinity at BS was almost identical between the two periods. Since rainfall was also very similar between periods (Figure 7), BS was considered to be a control site for salinity.

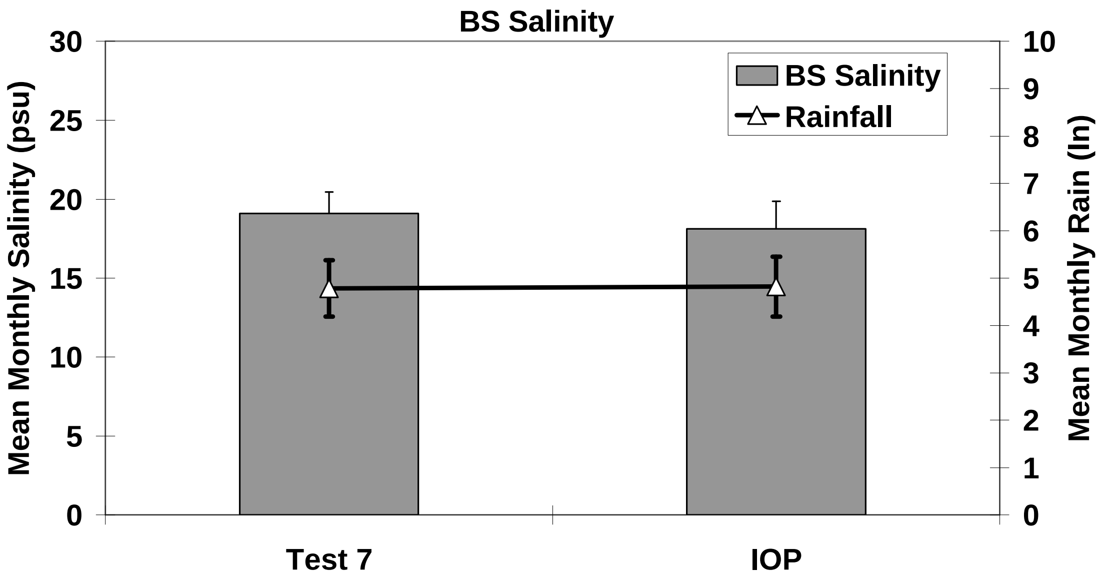

Figure 7. Mean rainfall and salinity at BS for each operational period. Two-Way ANOVA indicated no significant interaction between operational period and salinity or rainfall $F_{(1,140)}=0.19$; $p<0.6656$. Analyses were also performed and wet season and dry season separately, however, the results were similar to these. These results indicate that data collected at BS is a suitable control for BACI analyses.

BACI analysis was performed using rainfall and BS salinity as controls and TR, JB, and HC salinity used as impacted sites. Overall, there was no difference between Test 7 and IOP in annual mean salinity (Figure 8). However, when performed on individual seasons, BACI indicated significant difference (Figures 9 & 10). In both the wet and the dry seasons *post apriori* tests indicated there were no significant differences in salinity at the control sites, however, salinity was significantly higher under the IOP at the impacted sites (Figures 9 & 10). These results indicate that Test 7 was better than the IOP at achieving the desired responses of reduced salinity within coastal wetlands. Furthermore, based on the relationships between Taylor Slough flow and salinity in the mangrove wetlands (Appendix 2), these results suggest that Test 7 was better at forcing more water into the natural flowway of Taylor Slough. To illustrate this further, an examination of means and variance at each site (Figure 9) indicated that that the difference in salinity became more pronounced nearest the slough (TR, Figure 1) and decreased eastward as sites became more influenced by C-111 discharges (JB and HC, Figure 1). These site differences support the concept that Test 7 was better than IOP at distributing flows away from C-111 and into Taylor Slough.

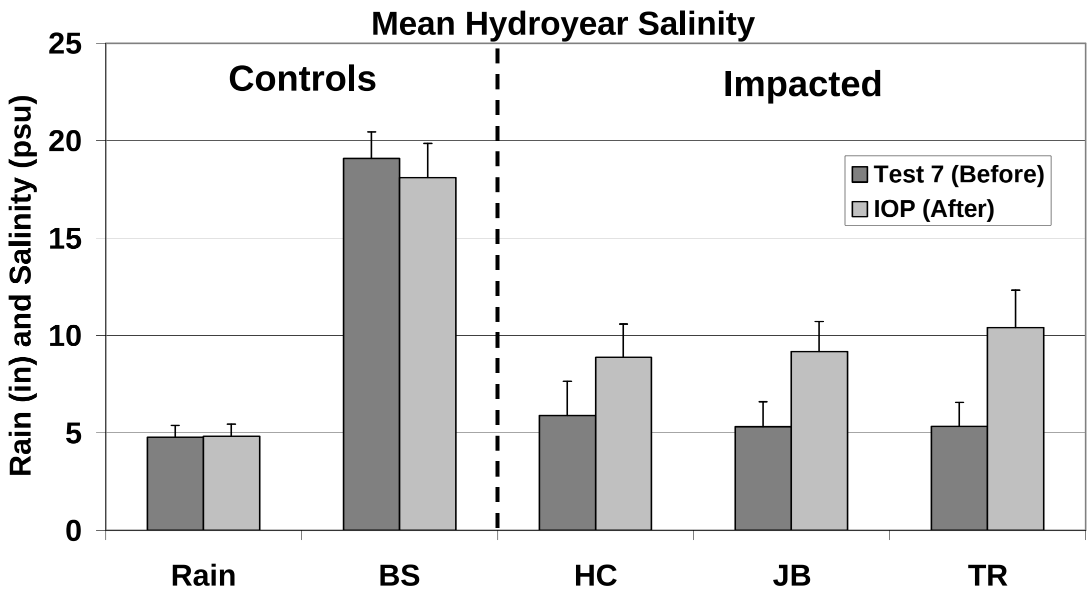

Figure 8. Mean rainfall and salinity (+se) at four sites for both operational periods. Analytical results indicate that the BACI interaction was not significant (AIC Model 1=26.4, AIC Model 2=25.1, ANODE=0.7), i.e, there was no significant difference between Test 7 and IOP at the control nor the impacted sites.

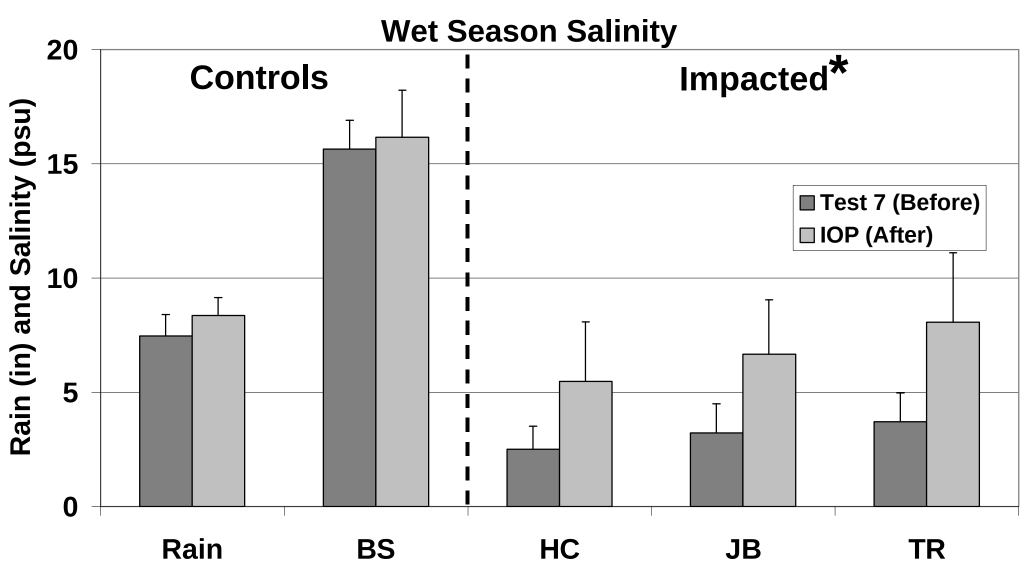

Figure 9. Wet season mean rainfall and salinity (+se) at four sites for both operational periods. Analytical results indicate that the BACI interaction was significant (AIC Model 1=-29.2, AIC Model 2=-26.6, ANODE=-2.6). Least squares means indicate that there was no significant difference in the controls but at the impacted sites, combined salinity was 113% greater during the IOP compared to Test 7 (*P=0.0186).

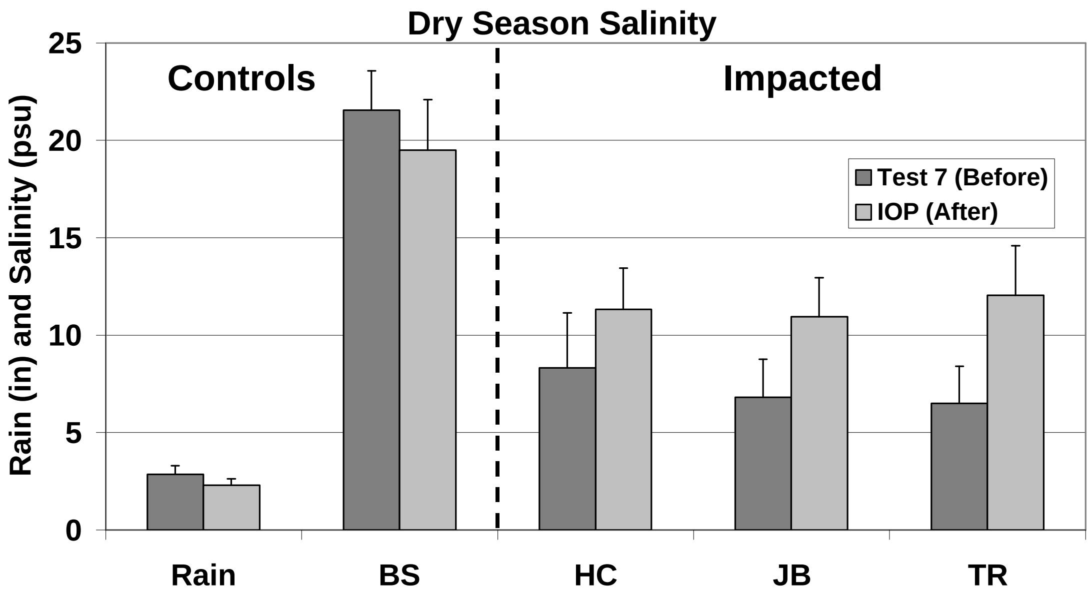

Figure 10. Dry season mean rainfall and salinity (+se) at four sites for both operational periods. Analytical results indicate that the BACI interaction was not significant (AIC Model 1=26.4, AIC Model 2=25, ANODE=1.4), i.e, there was no significant difference between Test 7 and IOP at the control nor the impacted sites.

Fish abundance was assessed using BS as the control site. Overall, there was no significant difference in the BACI analysis; however, the impacted sites had significantly more fish during Test 7 than IOP (Figure 11). When combined the impacted and control sites showed a clear statistical difference with Test 7 having higher fish abundance than the IOP (Figure 11). It may be biologically relevant (even though not statistically significant) that the difference at the impacted sites was greater (23%) than at the control (15%, Figure 11). The lack of a clear difference in this BACI may simply be a result of having too small a sample size from the control site: only eight control samples were collected each year because there was only one control site. In contrast, 24 fish impacted samples were collected each year because there were 3 impacted sites. Although not confirmed by the BACI analysis, these results suggest that there was a positive impact of Test 7 (or a negative impact of IOP) on fish abundance.

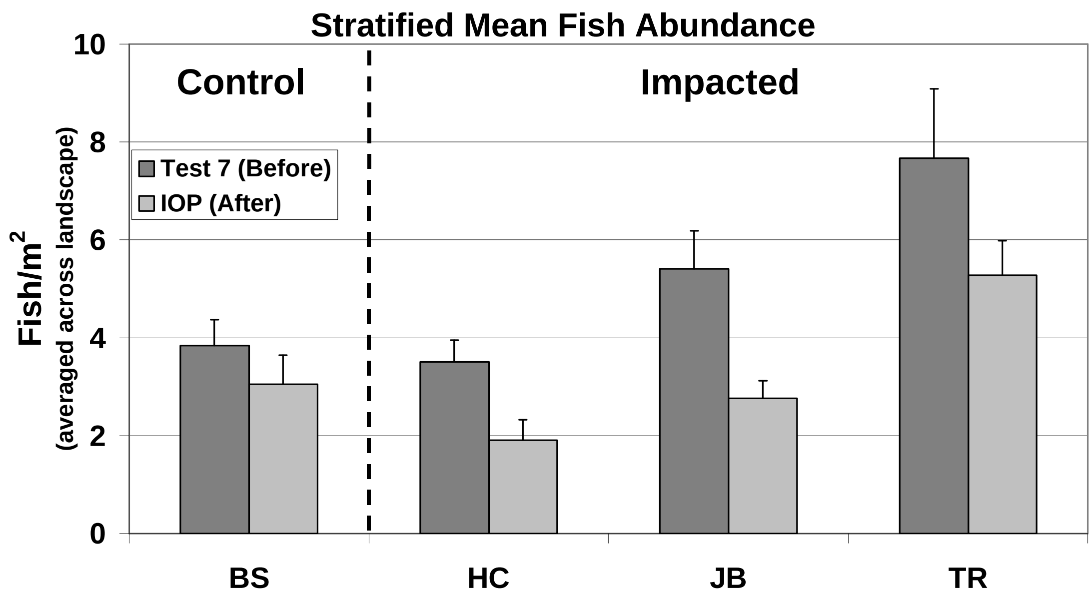

Figure 11. Stratified Mean Fish density (+se) at four sites for both operational periods. Analytical results indicate that the BACI interaction was not significant (AIC Model 1=408.6, AIC Model 2=409, ANODE=-0.4), i.e., there was no significant difference between Test 7 and IOP at the control nor the impacted sites. The BACI did indicate a Before and After effect, i.e., there was greater abundance under Test 7 than under IOP ($F_{1,175}$=9.58, P=0.0023). Since this was consistent at both the control and impacted sites, the higher abundance may be unrelated to the operational plan. However, least squares means indicated that the difference between plans at the control site (15%) was smaller than at the impacted sites, (23%) and that this difference at the impacted site was significant (p=0.0008).

Fish abundance is primarily determined by wet season salinity and hydroperiod and is pertinent in that higher abundance at the end of the wet season will result in greater fish availability during the dry season (Appendix 1). Therefore, BACI analysis was performed on fish abundance during the wet season only. As with the overall results, the BACI analysis indicated no significant difference (Figure 12). The timing of wet season fish collections may explain the lack of an affect when one was expected (based on salinity differences). The last sample collected during the wet season occurred in September, even though salinity remained low through January (Figure 13). This would suggest that, based on the salinity regime, the highest fish abundance should occur in January not September. However, there was a complicating factor in that water level dropped below the fish concentration threshold of 12.5 cm relative depth (the depth at which fish become concentrated and are subject to mortality events: see discussion below and Appendix 1) after December (Figure 13). Therefore, November and December were the last month of the hydrologic year with both low salinity and high water level and should be the peak months for fish abundance, i.e., a separate BACI was performed on fish abundance where November and December were included with the wet season. However, this analysis also indicated no significant difference in fish abundance where one was expected (Figure 14).

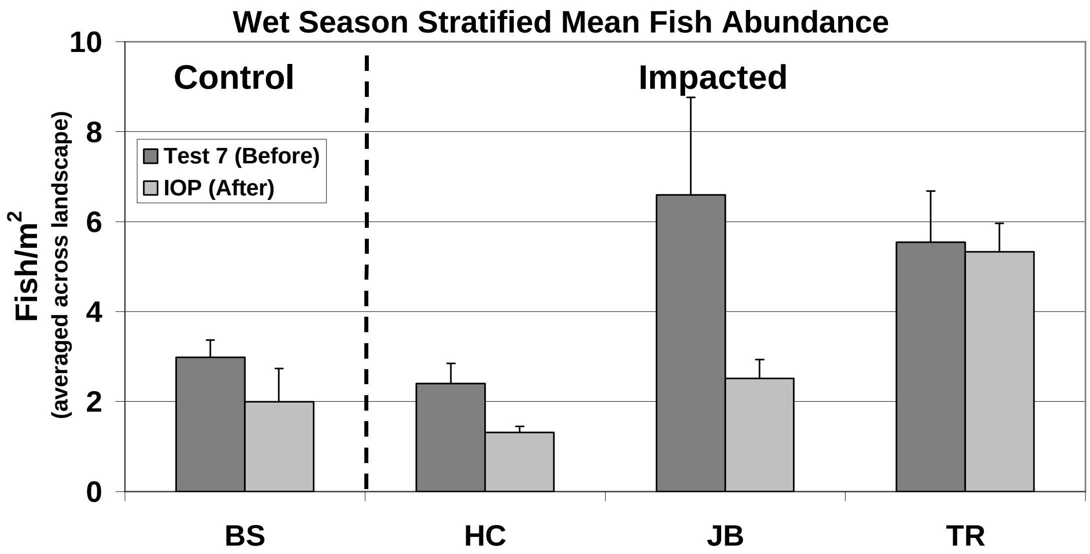

Figure 12. Wet season stratified Mean Fish density (+se) at four sites for both operational periods. Analytical results indicate that the BACI interaction was not significant (AIC Model 1=84.1, AIC Model 2=84.4, ANODE=-0.3), i.e, there was no significant difference between Test 7 and IOP at the control nor the impacted sites.

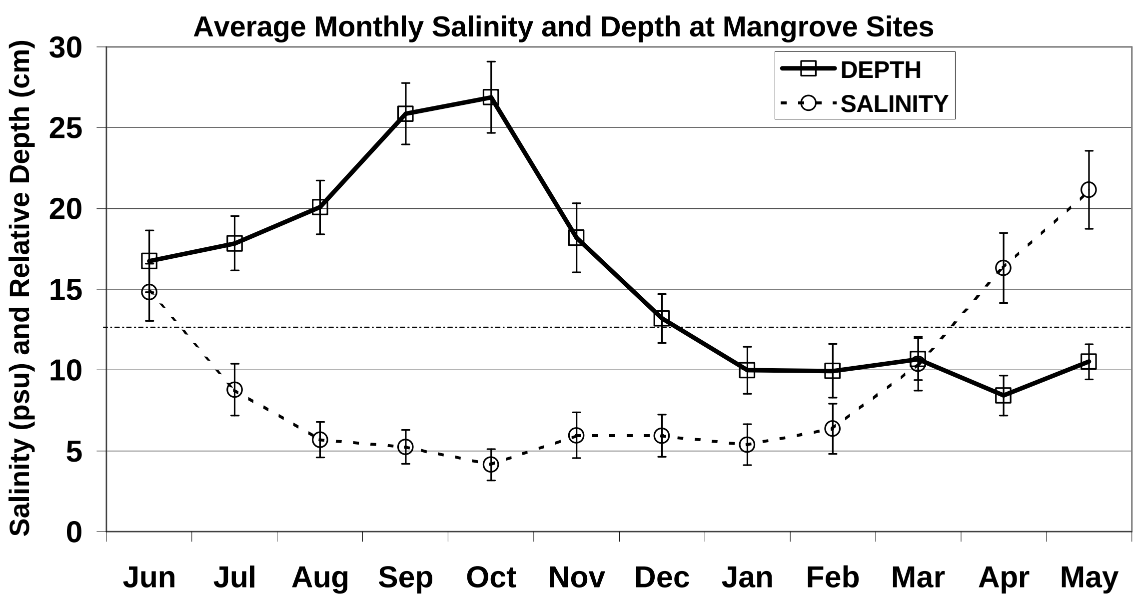

Figure 13. Average monthly (+se) salinity and depth from fish sampling sites

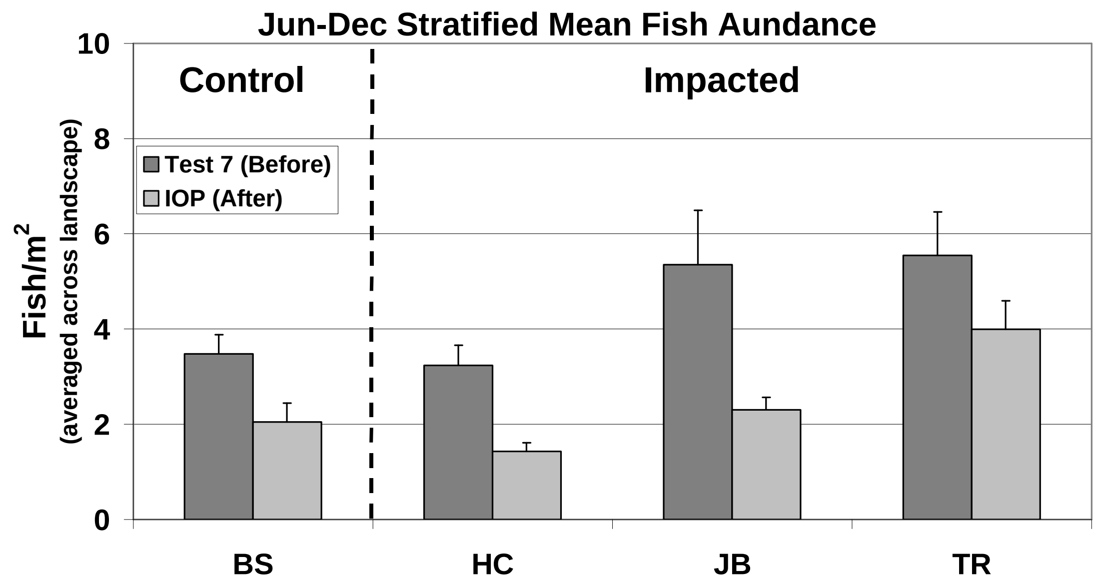

Figure 14. Stratified mean fish density (+se) calculated for June, September, November and December samples from four sites for both operational periods. Analytical results indicate that the BACI interaction was not significant (AIC Model 1=153.1, AIC Model 2=154.1, ANODE=-1), i.e, there was no significant difference between Test 7 and IOP at the control nor the impacted sites. However, the BACI did indicate a Before and After effect, i.e., there was greater abundance under Test 7 than under IOP ($F_{1,86}=19.07$, $P<0.0001$). Since this was consistent at both the control and impacted sites, the higher abundance may be unrelated to the operational plan.

Based on all the analyses of fish abundance, it is clear that there was a greater abundance of fish during the Test 7 period, however, there is no conclusive evidence that this was a product of water management operations. The results of the salinity analyses suggest that Test 7 was conducive to higher fish abundance and Figure 11 hints that higher abundance may have occurred, but was not significant because of lack of sample size in the control group. In contrast, Figures 12 & 14, suggest that there were more fish during the Test 7 period for reasons other than what can be explained by water management effects. The inconclusive nature of these results could be resolved by adding more control sites, and thereby increasing sample size.

4. Water Level Effects: Results and Discussion

As stated in the methods section, BS was subject to diurnal tides while the impacted sites were not. This resulted in differences in water level at BS that were not attributable to a rainfall driven system (Figure 15). To test the efficacy of using the BL site as a water level control, relative depth at BL was compared to rainfall (Figure 16). There was no statistical difference between the two periods indicating that BL could be used as a control site for water level.

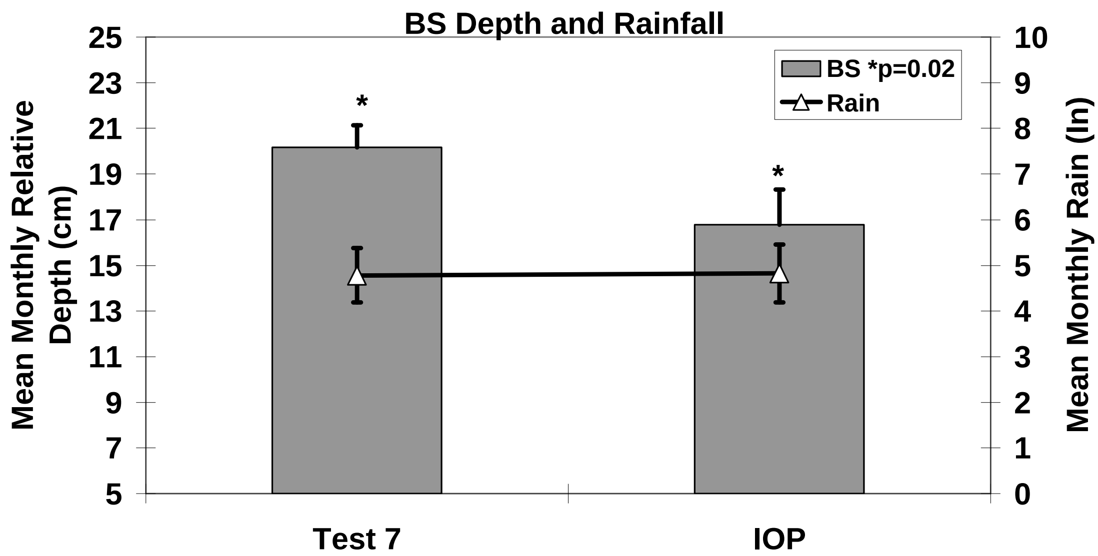

Figure 15. Mean rainfall and depth at BS for each operational period. Two-Way ANOVA indicated a significant interaction between operational period and salinity or rainfall ($F_{(1,140)}$=2.90; p<0.091). Least squares means indicated that rainfall was not different between operational plans but BS depth was significantly different (P=.02). These results suggest that data collected at BS is not a suitable control for BACI analyses.

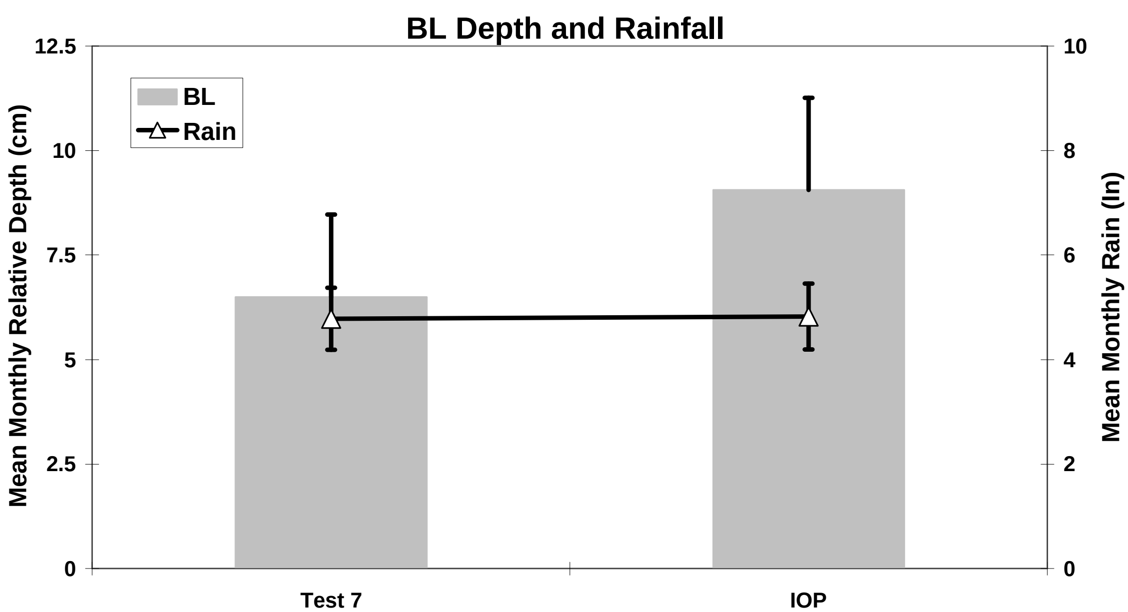

Figure 16. Mean rainfall and depth at BL for each operational period. Two-Way ANOVA indicated a significant interaction between operational period and salinity or rainfall ($F_{(1,126)}$=0.76; P<.3861). Least squares means indicated that neither rainfall nor BL Depth were different between operational plans (P>0.10). These results suggest that data collected at BL is a suitable control for BACI analyses.

BACI analysis was performed using rainfall and BL relative depth as controls and TR, JB, and HC relative depth as impacted sites. Overall, there was no apparent effect of water management on water level in the coastal wetlands (Figure 17). Although overall water level throughout the year is an important determinant of ecosystem function, the primary effect of depth on the system occurs during the dry season. Specifically, lower water levels are more desirable because they concentrate the prey base fishes (Appendix 1). Analysis of dry season water levels was significant with no difference between Test 7 and IOP in the controls but significantly higher water level occurred at the impacted sites during Test 7 (Figure 18). These results suggest that the IOP resulted in more desirable conditions in the dry season than Test 7.

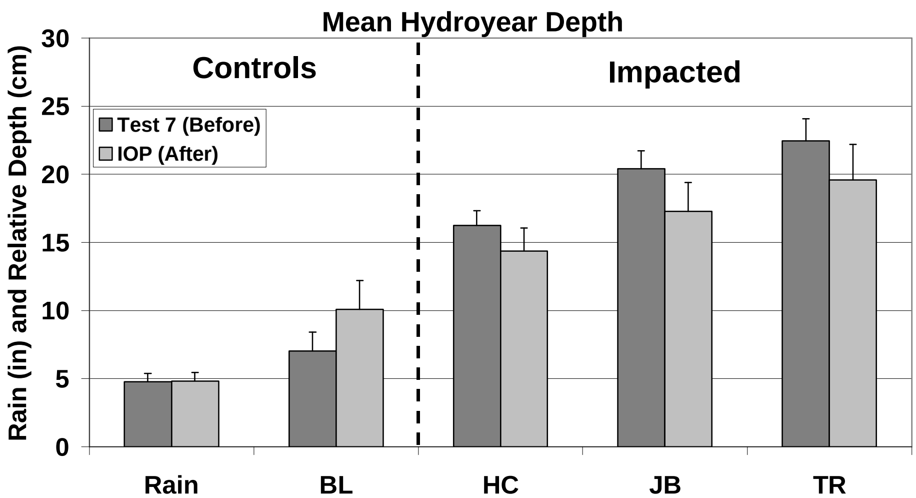

Figure 17. Mean rainfall and depth (+se) at four sites for both operational periods. Analytical results indicate that the BACI interaction was not significant (AIC Model 1=51.6, AIC Model 2=52.2, ANODE=-0.6), i.e, there was no significant difference between Test 7 and IOP at the control nor the impacted sites.

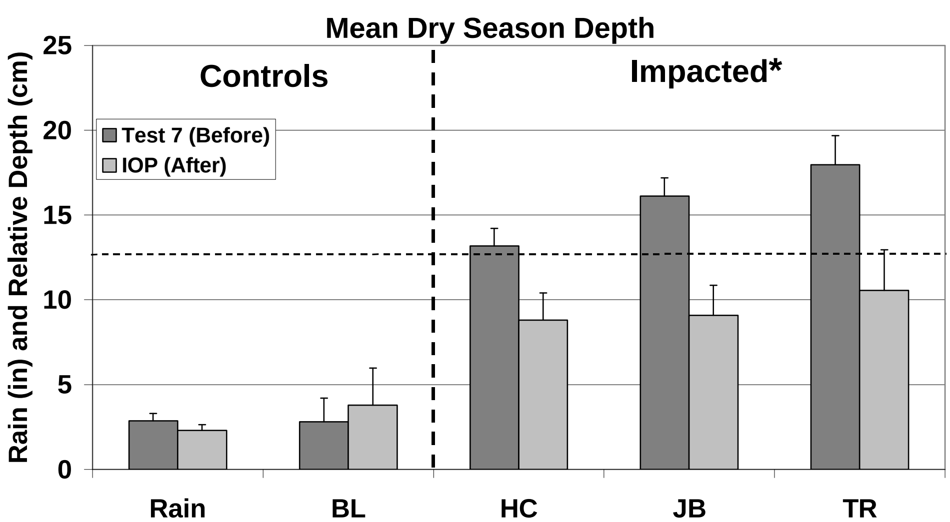

Figure 18. Dry season mean rainfall and depth (+se) at four sites for both operational periods. Analytical results indicate that the BACI interaction was significant (AIC Model 1=-111.4, AIC Model 2=-116.6, ANODE=5.2). Least squares means indicate that there was no difference in the controls but at the impacted sites, combined depth was 34% greater during the Test 7 compared to IOP (*P=0.0001). Also note that at the impacted sites mean water level was above 12.5 cm during Test 7 but below 12.5 cm during IOP.

An expected outcome of lower water levels during the dry season would be greater fish availability during IOP. The results of the BACI analysis of fish availability were significant (Figure 19) and support this expectation, however, the results require a more lengthy examination for this to be apparent. Figure 19 indicates that, at the control site, fish were more available during Test 7 than the IOP. It is unlikely that this difference was caused by better drying patterns (i.e., better concentration effect) during Test 7 since dry season water levels were about the same during both periods at the control site (Figure 18). A more likely explanation is that there were significantly more fish at the end of the wet season at the control site under Test 7 (Figure 11). This higher “starting” abundance at the beginning of the dry season resulted in greater concentrations of fish at the control site under Test 7 (Figure 19) even though water levels were about the same (Figure 18). Significantly higher abundance of fish also occurred at the end of the wet season under Test 7 at the impacted sites (Figure 11). If drying conditions were the same under both operational plans, higher availability would be expected under Test 7 at the impacted sites similar to what was observed at the control site. This was not the case: dry season fish availability was very similar for the two periods at the impacted sites (Figure 19). The most likely explanation is that the higher dry season water levels under Test 7 (Figure 18) masked the effect of the higher “starting” abundance at the end of the wet season: In other words; the lower water levels under IOP (Figure 18) resulted in fish being just as available to predators as under Test 7 (Figure 19), even though Test 7 had significantly higher fish abundance when calculated for the entire landscape (Figure 11).

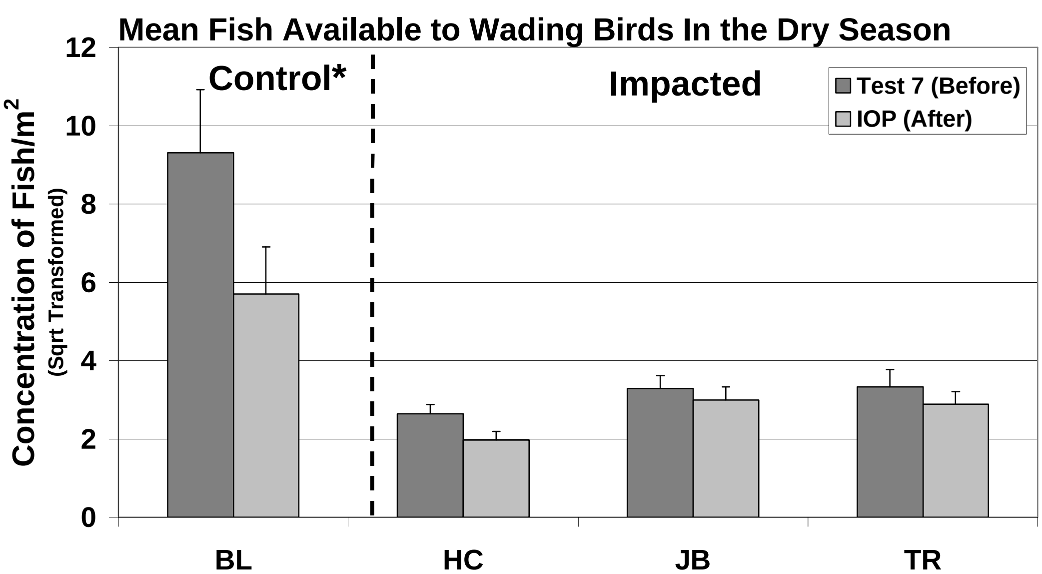

Figure 19. Mean fish number (+se) available to wading birds during the dry season (i.e., fish concentration) from four sites for both operational periods. (During Test 7, mean availability at BL was 131.5 +/- 33.5 Fish/m^2.). Analytical results indicate that the BACI interaction was significant (AIC Model 1=697.5, AIC Model 2=688.5, ANODE=9). Least squares means indicate that there was no significant difference at the impacted sites, however, at the control site, fish were 39% more concentrated during Test 7 than during the IOP (*P=0.004).

5. Impacts on Spoonbills: Results and Discussion

Spoonbills generally begin nesting in Florida Bay between November 1 and December 15 with a mean nest initiation date of November 18 (Alvear-Rodriguez 2001). The incubation period is approximately 21d. Once the eggs hatch, the chicks require constant care and an unbroken supply of food for about 42d. After this period, the chicks are more self-reliant but are still unable to leave the colony for another 42d. Based on these conditions, the critical period for spoonbills in Florida Bay is approximately from December 1 to March 31. Ideally, foraging spoonbills require that water levels remain at or below the concentration threshold of 12.5 cm (Appendix 1) somewhere within the coastal mangrove wetlands for the duration of this period.

BACI analysis indicated a significant impact of water management operations on water level for the spoonbill's critical period (Figure 20). Not surprisingly, these results were very similar to the conditions for the entire dry season (Figure 18) and indicated that Test 7 resulted in significantly higher water levels than the IOP at the impacted sites. More importantly, relative depth under Test 7 operations averaged well above the concentration threshold of 12.5 cm while the IOP resulted in water levels well below this stage (Figure 20). Under Test 7, relative depth at all three impacted sites averaged above 12.5 cm while the IOP resulted in mean depths below the concentration threshold (Figure 20). Fish availability for the spoonbill's critical period (Figure 21) was also very similar to that of the entire dry season (Figure 19) and should be interpreted the same way. In short, the IOP resulted in better foraging conditions for spoonbills in the coastal wetlands than Test 7.

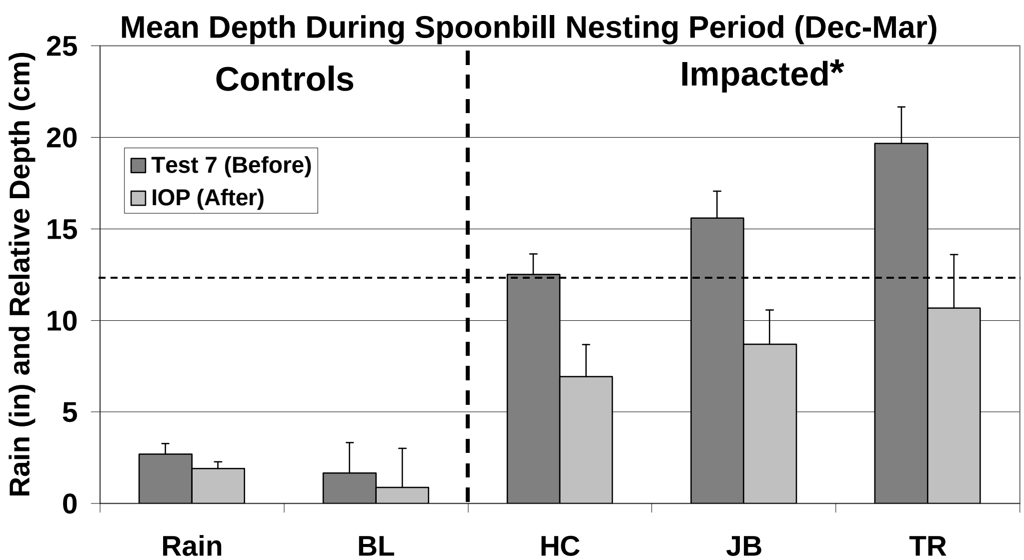

Figure 20. Mean rainfall and depth (+se) during the spoonbill-nesting season (Dec-Mar) at four sites for both operational periods. Analytical results indicate that the BACI interaction was significant (AIC Model 1=-111.4, AIC Model 2=-116.6, ANODE=5.2). Least squares means indicate that there was no difference in the controls but at the impacted sites, combined depth was 34% greater during the Test 7 compared to IOP (*P=0.0001). Also note that at the impacted sites mean water level was above 12.5 cm during Test 7 but below 12.5 cm during IOP.

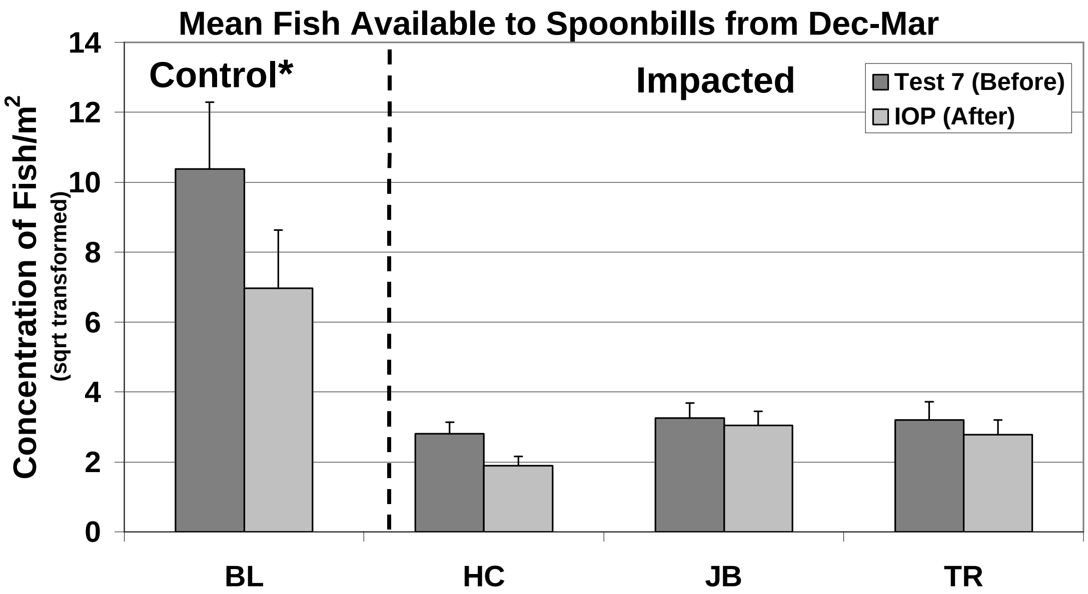

Figure 21. Mean fish number (+se) available (i.e., fish concentration) to spoonbills during their nesting season (Dec-Mar) from four sites for both operational periods. (During Test 7, mean availability at BL was 140 +/- 41.0 Fish/m^2.) Analytical results indicate that the BACI interaction was significant (AIC Model 1=455.6, AIC Model 2=448.9, ANODE=6.7). Least squares means indicate that there was no significant difference at the impacted sites, however, at the control site, fish were 34% more concentrated during Test 7 than during the IOP (*P=0.036).

Spoonbill nest production was also subjected to statistical analysis but the paucity of data points (three in each of the four categories) expectedly resulted in an inconclusive test (Figure 22). However, the data clearly indicate that Test 7 had a profound negative effect on spoonbill nesting success. At the impacted colony, fewer than 2 chicks fledged per 10 nesting attempts under Test 7. Under IOP, ten nests produced about 9 chicks. At the control site, the production rate was almost 8 chicks per ten nests during the Test 7 period and increased to 11 chicks per ten nests during the IOP. My interpretation of these data is that conditions throughout Florida Bay were not as good for spoonbills during the Test 7 period as they were during the IOP period. However, Test 7 operations exacerbated these relatively poor conditions causing almost complete failure in colonies impacted by these operations.

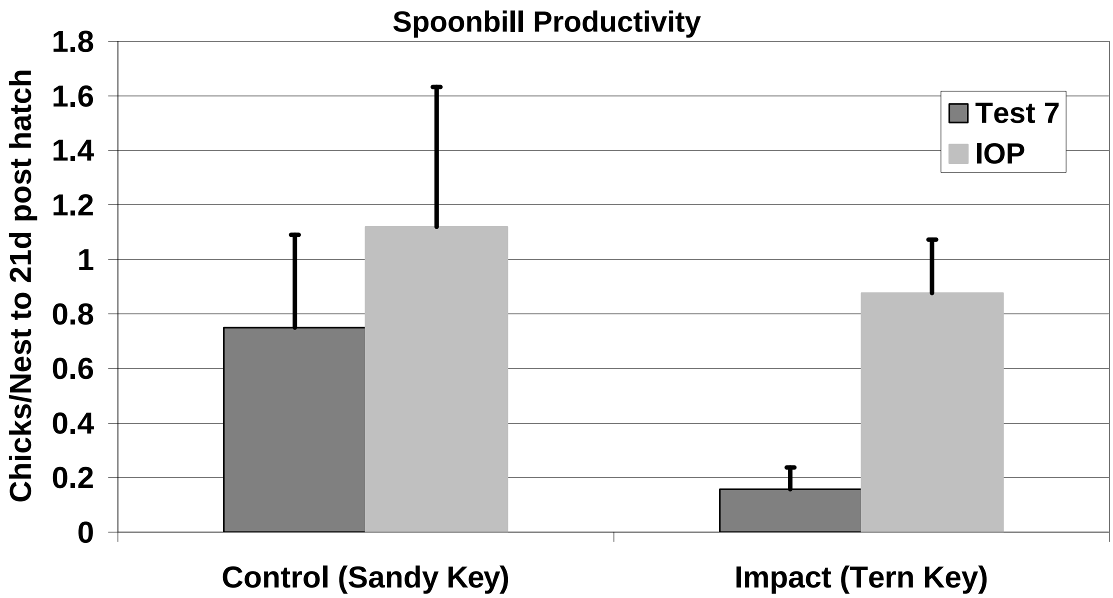

Figure 22. Mean success rate of nesting spoonbills at two locations for each operational plan. Two-Way ANOVA indicated no significant effect ($F_{1,8}$=.29; P<.6054), however, this was likely due to the small sample size (n=12 production estimates). From an ecological perspective, the virtual lack of reproduction at the impacted site under Test 7 and the more than 400% increase in production under IOP are indicative of anthropogenic effects. This is emphasized by the relatively high rates of production under both Test 7 and IOP at the unimpacted location.

6. Conclusions

The comparison of IOP to Test 7 had mixed results in achieving "desirable" conditions in the coastal wetlands. Test 7 appeared to be a better operating plan during the wet season while IOP performed better during the dry season.

Flow data suggest that Test 7 was superior to IOP in distributing water flow toward Taylor Slough and away from C-111 during the wet season. This may explain the observed differences in salinity at the impacted sites. Test 7 clearly resulted in lower salinity than IOP in the coastal mangrove zone. This was considered desirable because it presumably resulted in higher fish productivity (Appendix 1). The fish data collected during the study period do not conclusively confirm this presumption, however, the results do strongly suggest that such an effect occurred. A larger sample size within the control group may have provided more conclusive results.

IOP resulted in lower water levels during the dry season. The most conspicuous biological effect of lower water level is that the drying wetland concentrates prey into small pools thereby making them available to myriad predators (of which spoonbills are just one example). IOP appeared to reduce dry season pulse releases to the C-111 basin presumably reducing the number and extent of episodic reversals in the drying pattern. These conclusions are most strongly supported by the observed lower water levels in the impacted coastal wetlands. Furthermore, water levels were more consistently below the point where prey fish become concentrated under IOP than Test 7. This was particularly true during the spoonbill's critical nesting period, a likely explanation for why spoonbills had relatively good nesting performance under IOP but very poor nesting success under Test 7.

7. Recommendations

1. Complete the Modified Water Deliveries Project so that more natural flows and water levels are restored in Northeast Shark Slough. This would have downstream effects by re-flooding the Rocky Glades and allow recharge of the Taylor Slough Headwaters. In the context of this report, this would stimulate flow toward the central part of Florida Bay thereby lowering salinity and increasing biological productivity in the coastal wetlands.

2. Reestablish a ratio of flows whereby two thirds of the total flow toward Florida Bay is delivered via Taylor Slough and one-third is delivered to the Panhandle Region. Although Recommendation 1 would go a long way toward accomplishing this recommendation, operation of the L-31N/C-111 complex of canals, structures, pumps, and retention areas will need to be evaluated and updated to accomplish this goal. This effort should be included as a goal in the Combined Structural and Operational Plan (CSOP) which will replace the IOP when projects are completed.

3. Immediately raise water levels in the L-31N/C-111 complex of canals to Test 7 Phase 2 levels and standards so as to reduce seepage and increase flow control. This is especially true in the Lower C-111 where most of the flow through S-18C is a result of seepage from the Taylor Slough area. Canal levels should remain above ***x*** ft downstream of S-18C until such a time as Recommendation 4 is completed.

4. Complete the C-111 Project as planned such that the new C-111 spreader canal is extended under US-1. This will move the discharge point of the canal farther north, thereby buffering pulse releases to the coastal wetlands. Furthermore, the lower reach of the C-111 must be backfilled so as to prevent any further temptation to shunt unwanted water through S-197 thereby causing salinity stress in Barnes Sound with the unintended consequence of raising water in the coastal wetlands. The entire existing canal below where the new spreader canal is put in place must be backfilled or the extraordinarily destructive effect of dry season reversals will continue.

8. Monitoring Needs (in order of importance)

1. Additional control sites need to be established east of US-1 in the Barnes Sound and Card Sound areas. This would solve the statistical problem of small sample sizes realized in the analyses of fish abundance. Two new sites were identified

in 2001 and modest sampling has occurred at these locations, however, funding beyond 2003 is unlikely from the current resource (the South Florida Water Management District). These sites should be incorporated into the existing project and funded under the existing agreement with the Corps and ENP.

2. Even with the additional control sites, the project suffers statistically from a lack of replicates. Each site has unique hydrologic conditions and, as such, can not be considered replicates. This problem can be solved by placing an additional sampling location near the existing sites such that paired samples are collected. This would greatly increase the statistical power of this monitoring program.

3. Establish monitoring of invertebrates in conjunction with the fish sampling sites. Other parts of this report clearly indicate that invertebrates have been shown to be good indicators of ecosystem conditions. This is particularly true in deducing the effects of salinity. Adding this component to the existing monitoring program would greatly enhance the scope and sensitivity of the research.

4. Further refine the physical models so that the predictive models developed for fish and spoonbills can be used to assess existing and planed operations through simulation. The existing models need to be become more precise as well as accurate before these models can be fully used.

5. Physical monitoring of the southern Cape Sable region plus adding an additional fish sampling site would allow for increased options in data analysis. Although adding sites East of US-1 is most helpful (Monitoring Need #1), these sites will be influenced by future water management activities. Cape Sable provides a true control in that it spatially distant from these activities. Ultimately, the area from Flamingo to East Cape Sable will provide us the only possible location for CERP control sites since CERP is designed to directly impact almost all other regions.

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APPENDIX 1: BACKGROUND INFORMATION AND PREVIOUS FINDINGS

The ecotone between the freshwater Everglades and the marine environment of northeastern Florida Bay is a relatively narrow (= 5 km) band of coastal mangrove forest. Receiving influences from both freshwater and marine systems, this mangrove ecotone has the capacity to reflect ambient conditions in both major ecosystems and, is therefore, an ideal habitat for evaluating anthropogenic impacts on the larger hydroscape. The primary goal of the MERP is to investigate the impact of upstream water management practices in the Taylor Slough and C-111 basins on the downstream mangrove ecosystem. To this end, the MERP has established four field locations within the mangrove ecotone at which physical and biological parameters are sampled using scientific methodologies. Salinity and water level data are the principal parameters used to evaluate the physical environment while aspects of the resident demersal fish community are used to evaluate within ecosystem function. This fish community is the primary prey base for myriad predators (e.g. piscivorous birds, fishes, reptiles and amphibians). Roseate Spoonbills are a conspicuous bird species that depend on this prey source in order to nest successfully in nearby northeastern Florida Bay (Lorenz et al. 2002). Therefore, Roseate Spoonbill nesting parameters are also monitored as part of the MERP so as to evaluate the linkages between biotic function with the mangrove ecotone and the surrounding hydroscape.

Importance of mangrove fishes. The community of small fishes that thrive in the mangrove areas north of Florida Bay are a vital food source for a variety of important animal species. In addition, these prey base fishes are excellent indicators of ecosystem health because they have naturally high rates of reproduction and mortality (their life span can be as short as 3 months). Consequently, the community will quickly show quantifiable changes in response to changes in habitat quality. These fishes are also useful as indicators because their small size limits movement. As a result, they spend their entire life cycle within the mangrove habitat and are termed resident mangrove fishes. In contrast, many of the larger fish species, birds and reptiles that are used as indicators of ecosystem health are wide ranging. This allows them to move to more favorable secondary habitats when conditions deteriorate in their primary habitat. These mangrove transients may not accurately reflect short term perturbations in the mangrove system. Since the resident fish community cannot leave the system when adverse conditions arise they better reflect changes in mangrove hydrology.

Description of fish sampling sites General areas for site locations were selected based on the potential to show differences in the prey base community due to the influence of natural fresh water Everglades sheet flow. Specific sampling sites were selected based on close proximity (within 1 km) to ENP hydrological monitoring stations so that the data collected by these stations could be used to interpret the results of fish collections. All sites were located in similar dwarf mangrove habitat and each was characterized by a deep central creek surrounded by shallow flats that became seasonally exposed. Vegetation consisted of widely spaced (0.5-5.0 m between plants) dwarf red mangrove (Rhizophora mangle) trees (0.5-2.0 m tall) with varying amounts of vegetation between trees. Growth of Eleocharis sp., Utricularia sp. and Chara sp. was seasonal. The substrate for all sites was flocculent, unconsolidated, carbonate marl (Browder et al. 1994).

Hydrologic cycles. Figures 13 of the main report provides an example of the annual cycles in salinity and water level. The mangrove ecotone of northeastern Florida Bay does not experience a lunar or diurnal tide (Holmquist et al. 1989) so the annual cycle of water depth change in the mangrove swamp is controlled by three factors that cycle seasonally: sea level, wind, and rainfall. Starting approximately in May or June, water levels climb throughout the summer months and peak in October. Typically, water levels rapidly decline through November and December culminating in dry season conditions from January through April or May.

Salinity follows a similar but inverted pattern to water level. Salt concentrations are typically highest in late May and rapidly decline with the onset of the wet season in June. By August nearly freshwater conditions are achieved. Salinity remains low throughout the wet season and is typically at or near freshwater conditions from September through January. Beginning in February salinity rises rapidly and peaks in May. Because the natural break point in the annual hydro-cycle is the initiation of the wet season, all analyses focus on the hydrologic year (June to May). These stereotypic cycles are affected by a high degree of spatial, inter-annual and intra-annual variation of physical conditions in the mangrove zone.

Results from previous fish collections. Several studies of fishes in southern Florida freshwater wetlands have demonstrated that seasonal fluctuations in water level have a dramatic effect on the fish community (Higer and Kolipinski 1967; Kushlan 1980, Duever et al. 1986, Loftus and Kushlan 1987, Loftus and Eklund 1994, DeAngelis et al. 1997, Lorenz 2000). The paradigm (Figure A1.1) postulated by these studies was that during high water periods, fishes utilize expansive ephemeral wetlands as foraging grounds and refugia from predation (Kushlan 1980, DeAngelis et al. 1997, Lorenz 2000). This exploitation of periodically flooded areas results in exponential increases in fish abundance (Loftus and Eklund 1994, DeAngelis et al. 1997). During the dry season, wetland surfaces become dry and fishes either become stranded or are forced into deeper areas (Kushlan 1978, Kushlan 1980, Loftus and Kushlan 1987, Loftus and Eklund 1994, DeAngelis et al 1997, Lorenz 2000). This "concentration effect" exposes the wetland fishes to high rates of mortality due to strandings, adverse physical conditions (e.g. low dissolved oxygen and high temperatures) and heavy predation from a variety of piscine and avian predators (Hunt 1952**,** Kushlan 1978, Master 1989, Master 1992, DeAngelis et al., 1997, Lorenz 2000).

Fish collections made between June 1991 and May 1999 were analyzed in conjunction with hydrologic data (Lorenz 1999, Lorenz 2000). Analyses focused on the impact of fluctuating water level and salinity on the fish community in order to better understand the impact that flow manipulation has had on the ecotone. The data strongly supported the above paradigm. In the freshwater Everglades, Kushlan (1976b) indicated that prey fish start to become concentrated in pond habitats at about 10-20 cm depth relative to the depth at which the surrounding wetland became dry. Lorenz (2000) performed a more rigorous analysis of the depth/fish concentration relationship at the four mangrove sites and found that fish begin to be forced from the wetland surface and become concentrated in the deeper creek habitat at about 12.5 cm relative to the deepest part of the flats (Figure A1.1).

Changes in biomass were primarily related to long term salinity conditions and secondarily to long term changes in water level. Analysis of variance of biomass between sites indicated that sites with longer freshwater periods had higher biomass than sites with shorter freshwater periods. Detrended Correspondence Analysis (DCA) of community biomass supported the hypothesis that salinity was a primary determinant of community structure and that community structure was a primary determinant of fish standing stock (Lorenz 1997, Lorenz 2000, Lorenz and Serafy in prep)

These results indicated that anthropogenic changes in water delivery could have negatively impacted the ecotonal prey base. The implication is that decreased freshwater flow to Florida Bay may have eroded the ecosystems trophic structure from the bottom up. This situation is potentially reversible: changes in the quantity, timing and distribution of fresh water deliveries to the mangrove zone via Taylor Slough and the C-111 canal could create longer hydroperiods in the mangrove zone. Longer hydroperiods would also reduce salt water intrusion into the area, and may help re-create the higher secondary productivity that was once associated with the Everglades/Florida Bay interface.

Results from Roseate Spoonbill studies. Surveys of Florida Bay's Roseate Spoonbill nesting population conducted since 1950, indicated that spoonbills relocated nesting effort and experienced reduced nesting success in response to anthropogenic perturbations to their foraging grounds (Lorenz et al. 2002). Birds nesting in northeastern Florida Bay primarily foraged in the coastal wetlands from western Taylor Slough to Turkey Point. Starting in the mid-1980s the number of nests in the northeastern basin began a steady decline. In 1999, less than 22% of Florida Bay's spoonbills nested in these colonies, down from the approximately 60% that nested between 1967 and 1982. Nesting success in the northeastern colonies decreased from an average of 1.4 chicks per nest between 1966 and 1982 to an average of less than 0.7 after 1984. Concurrent with this decline in the northeastern colonies, the percentage of nests in the northwestern region of the bay increased from less than 20% prior to 1984 to 53% in 1999. Likewise, the number of nests found in the central region increased from only a few nests per year to more than 10% of the total nests. The average success of these colonies from 1984 to 1999 was 1.2 chicks per nest. The decline in nesting effort and nesting success in the northeastern bay and the shift to nesting in other regions indicated a degradation of the foraging grounds in the coastal wetlands associated with Taylor Slough and the C-111 basin (Lorenz et al. 2002).

As mentioned above, analyses of the prey-base fish community in these wetlands indicated that water management practices since the mid-1980s have adversely impacted the abundance and availability of prey species for spoonbills. More specifically, Lorenz (2000) demonstrated that when spoonbill nesting failed, water levels on the foraging grounds were above the concentration threshold of 12.5 cm (Figure A1.1) and prey fish were not concentrated (i.e., low availability). During successful nestings, water levels remained below 12 cm during the critical nesting period and fish were highly available (Figure A1.1). These results indicated that Roseate Spoonbills respond in a predictable manner to impacts caused by changes in water management and are, therefore, a good indicator species for the overall health of the Florida Bay estuary. Changes in water management designed to restore more historic hydrologic regimes should improve conditions for spoonbills by making prey both more abundant and more available. Therefore, water management activities can be evaluated based on spoonbill responses.

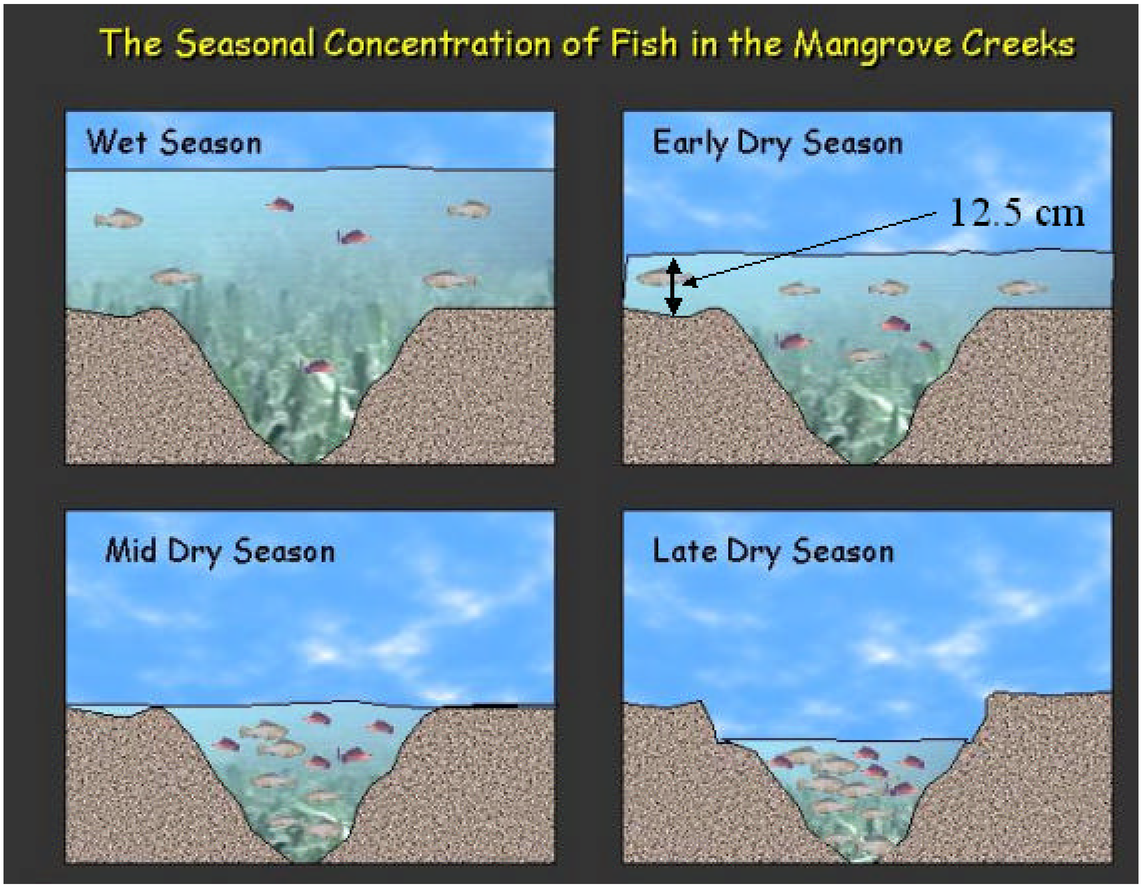
The Seasonal Concentration of Fish in the Mangrove Creeks
Wet Season
Early Dry Season
12.5 cm
Mid Dry Season
Late Dry Season

Figure A1.1

Appendix 2. A reconstruction of wetland water level and salinity patterns in NEFB using historical data, hydrologic models and the hydrologic requirements of an umbrella-type indicator species

Introduction

Surveys of Florida Bay's Roseate Spoonbill nesting population conducted since 1950, indicate that spoonbills relocate nesting effort and experience reduced nesting success in response to anthropogenic perturbations to their foraging grounds (Lorenz et al. 2002). Birds nesting in northeastern Florida Bay primarily forage in the coastal wetlands from western Taylor Slough to southern Card Sound (Figure A2.1; Bjork and Powell 1994, Lorenz et al. 2002). Starting in the mid-1980s the number of nests in the northeastern basin began a steady decline. In 1999, less than 22% of Florida Bay's spoonbills nested in these colonies, down from the approximately 60% that nested between 1967 and 1982. Nesting success in the northeastern colonies had decreased from an average of 1.4 chicks per nest between 1966 and 1982 to an average of less than 0.7 after 1984 (Lorenz et al 2002). Concurrent with this decline in the northeastern colonies, the percentage of nests in the three colonies of the northwestern region of the bay increased from less than 20% prior to 1984 to 53% in 1999 (Lorenz et al 2002). Likewise the number of nests found in the four colonies from the central region increased from only a few nests per year to about 10% of the total nests. The average success of these colonies from 1984 to 1999 was 1.2 chicks per nest. The decline in nesting effort and nesting success in the northeastern bay and the shift to nesting in other regions indicates a degradation of the foraging grounds in the coastal wetlands associated with Taylor Slough and the C-111 basin (Lorenz 2000). A study of the fish community in these wetlands indicated that water management practices since the mid-1980s have adversely impacted the abundance and availability of prey species for spoonbills (Lorenz 1999). These results indicate that Roseate Spoonbills respond in a predictable manner to impacts caused by changes in upstream water management (Lorenz 2000, Lorenz et al. 2002).

In the case of northeastern Florida Bay and associated coastal wetlands, Roseate Spoonbills appear to be representative of other species of interest. A variety of reptilian, piscine, and avian predators utilize the same resources as spoonbills. Several species of turtles and snakes known to eat fish utilize the mangrove areas (Odum et al. 1982, pers. obs.) and young crocodilians (a federally protected species) utilize this area during their critical post-hatching period (Mazzotti 1999). Piscivorous fishes including snook, tarpon, crevalle jack, gray snapper and spotted gar were frequently seen hunting at foraging sites (Faunce et al. 2003). Wood Storks, Great Egrets, Snowy Egrets, Tricolor Herons, Reddish Egrets, White Ibis, Great White Herons and Great Blue Herons were observed foraging in the coastal wetlands during periods of high fish concentration (Powell 1987, Bjork and Powell 1994). Like Roseate Spoonbills, many of these species time reproduction to correspond to low water levels and high fish concentration (Powell 1987, Powell et al. 1989). Also like spoonbills, many of these species have exhibited declines in numbers over the last several decades (Lorenz 2000). Based on the ability to document the responses of spoonbills to perturbations and that many other species are dependent on the same resources, Roseate Spoonbills are an excellent candidate for an indicator of ecosystem health for this estuary.

Studies of the demersal fish community in these wetlands indicated that water management practices since the mid-1980s have adversely impacted the abundance and availability of the spoonbill's primary prey (Lorenz 2000). Analyses yielded two major effects of upstream water management practices on the spoonbill/prey fish relationships. The first was that diversion of natural flows resulted in alterations of the salinity regime in the mangrove zone (McIvor et al. 1994, Lorenz 2000). The change in salinity patterns negatively effected primary production in the submerged aquatic vegetation (SAV) community within the mangrove zone (Montague and Ley 1993, Frezza et al. 2002) which, in turn, resulted in lower abundance of prey base fishes (Ley et al. 1994, Lorenz 1997, Lorenz 2000). The observed decline in productivity within the prey base fishes likely explains (at least in part) declines in predator populations dependant on this food resource (Bancroft et al. 1994, Lorenz et al. 2002). The second major effect of water management was that pulse releases of water from the canal system during the dry season resulted in reversals of the seasonal drying patterns within the coastal mangrove ecosystem. Drying events are critical to the ecosystem in that myriad predators take advantage of a highly abundant food source in the form of prey base fishes concentrated into the remaining small pools (Kushlan 1976a, Kushlan 1976b, Loftus and Kushlan 1987, Loftus and Eklund 1994, Lorenz 2000). Spoonbills and many other wading bird species gear the timing of nesting to these drying events which enables them to readily meet the high energetic requirements of their rapidly growing young (Frederick and Colopy 1989, Bjork and Powell 1994, Lorenz 2000, Dumas 2000). Anthropogenic reversals in the drying pattern allow prey base fishes to spread out across the landscape making them relatively unavailable to wading birds, thereby resulting in nesting failure (Lorenz 2000). The critical depth at which fish begin to move into deeper habitats (i.e., begin to concentrate) is when water levels on the wetland surface drops below 12.5 cm or 5 inches (Lorenz 2000). Lorenz (2002) also showed that if water levels exceed this threshold during spoonbill nesting period then nesting fails.

Based on the above findings, two performance measures regarding ecosystem function in the coastal wetlands north of eastern Florida Bay were developed. 1) That salinity in the coastal wetlands should remain consistently low (<5psu) through the early dry season (January) so as to create ideal conditions for high prey fish productivity and; 2) that water levels somewhere within the spoonbill foraging range (as defined above) should be less than 12.5 cm in depth from December through at least the end of March.

Prior to the establishment of Everglades National Park, spoonbills were largely extirpated from Florida first as part of the plume hunting trade and then through subsistence hunting (Allen 1942, Powell et al 1989). Once protected, spoonbills recolonize southern Florida Bay near the Florida Keys (Allen 1942, Allen 1947) and expanded outward from this starting point (Powell et al 1989, Lorenz et al. 2002). Anecdotal evidence from the 1950’s suggests that northeastern Florida Bay was a major nesting and foraging area for spoonbills prior to the direct water management impacts to the region that occurred in the 1970’s and 1980’s. Flight line data from many colonies throughout the southern bay indicated that the mainland was being used significantly as a foraging area even though it was a relatively long flight (Robert Porter Allen personal notes, data presented in Lorenz et al. 2002). This indicates that the foraging quality of these wetlands was relatively high compared to other available foraging grounds in the Florida Bay area. Initial expansion of nesting into the northeastern colonies from the southern colonies began prior to 1950 and accounted for 10 to 30% of the total nests in Florida Bay between 1950 and 1963, thereby pre-dating both water management impacts in the Taylor Slough region and loss of foraging habitat on the mainline Keys (Lorenz et al. 2002). Estimates of nesting success in the northeastern colonies were made in only two years prior to water management impacts (2.3 chicks per nest were recorded in 1955-56 and 3.0 in 1961-62) but these estimates were much higher than what occurred following water management impacts (the mean chick per nest from 1970 to 1982 was 1.2 with a maximum 2.2). Although only limited flight line data was collected for a northeastern colony during the 1955-56 and 1961-62 nesting cycles, these limited observations show that spoonbills were primarily foraging on the mainland.

Collectively, these observations indicate that, prior to the water management impacts, the quality of forage habitat on the mainland wetlands was high enough to stimulate a significant portion of Florida Bay's spoonbills to forage in the area, to select nest sites in the northeastern colonies, and to have a high degree of success in so doing. Given the requirements of spoonbills as developed above, one might conclude that prior to direct water management effects, the coastal wetlands met the specific foraging needs of spoonbills during their nesting cycle. More specifically, I hypothesize that the historical condition of these wetlands met the performance measures of low salinity (<5 psu) and low water levels (<12.5 cm somewhere within foraging area) through March.

On the surface, these two performance measures appear contradictory in that drawing down the water levels on these wetlands would allow marine intrusion and increased salinity. However, this seeming paradox can be resolved by an investigation of current water delivery patterns to the coastal wetlands and how freshwater may have arrived prior to water management practices. Although no consistent hydrologic or hydrographic data is available for this region prior to 1990, hydrologic models have been developed which can be used to hindcast these data. Hydrologic models have been used to assess the effect of both historic and future effects of water management practices in the Everglades (Fennema et al 1994, DeAngelis et al. 1997, Sklar et al.

2002). These models attempt to estimate flows of freshwater to Florida Bay via the Taylor Slough and Panhandle drainage basins. I will attempt to test the above hypotheses using existing conditions, model generated historic data, and the sparse historic data that exists for the region.

Historical flow patterns within the Taylor Slough/Northeastern Basin watershed

The Taylor Slough headwaters region is a shallow, poorly defined depression situated between the upland habitats of the Pineland Ridge to the south and east and a lower lying rock ridge called the Rocky Glades to the north and west (Figure A2.2; Craighead 1971, Schomer and Drew 1982, Johnson and Fennema 1989). In the pristine Everglades, the primary source of water for Taylor Slough was Shark River Slough (SRS) with runoff from the pineland coastal ridge making a significant contribution to the water budget (Schomer and Drew 1982, McIvor et al. 1994). Prior to urban development in southern Florida, SRS was widest approximately 5 km north of US Highway 41 (Figure A2.2). At this point, the slough was approximately 46 km across on an east to west transect (Davis 1994). South of US 41, the Rocky Glades slope southwestward forcing the course of SRS in that direction (Craighead 1971, Van Lent et al. 1993, Davis 1994). On the west side of SRS, the Cypress Ridge restricts the flow of water to the west (Craighead 1971, Davis 1994). As the Cypress Ridge and the Rocky Glades converge to the south, they act as a funnel and SRS shrinks to less than 7 km across (Figure A2.2; Davis 1994). This constriction causes water to pond in the upper reaches on the eastern side of SRS in an area called Northeastern Shark Slough (NESS; Fennema et al. 1994).

Prior to anthropogenic drainage efforts (i.e., before 1900), during a typical wet season, water levels would rise in NESS until the Rocky Glades were over-topped and water would sheet flow south and east into the Taylor Slough headwaters basin (Fennema et al. 1994, USACOE 1994). As the Taylor Slough headwaters basin began to fill, water would be funneled southward by the Pineland Ridge toward Paradise Key (Figure A2.2) where it would exit the basin through large gaps in the ridge (Schomer and Drew 1982, USACOE 1994). These gaps in the Pineland Ridge are the beginning of Taylor Slough proper (Figure A2.3; Schomer and Drew 1982). About 7 km south of Paradise Key is a shallow rise first identified by Tabb et al. (1967, Figure A2.3). This rocky area is about

30 cm higher in elevation than the surrounding wetlands (Ley et al. 1995) and shares a geological kinship with the Rocky Glades (Tabb, Tropical Bioindustries 1990). Tabb's Rise (Figure A2.3) diverts the Taylor Slough flow way to the west. East of Paradise Key, several smaller breaks in the Pineland Ridge called Finger Glades or Transverse Glades (Figure A2.3) allowed water to exit the headwaters basin toward the southeast (Craighead 1971, Schomer and Drew 1982, Ley et al. 1995, Simmons and Ogden 1998). Flow from the Finger Glades created smaller sloughs (e.g. Loveland Slough, Winkeye Slough) that flowed southeastward from the Pineland Ridge. Large amounts of fresh water were delivered to northeastern Florida Bay (eastern half of Joe Bay eastward to Long Sound) and to Southern Biscayne Bay (Manatee Bay, Barnes Sound and Card Sound) via these water ways (Craighead 1971, Schomer and Drew 1982, Simmons and Ogden 1998). Tabb's Rise separates these two distinct freshwater flow ways from the Headwaters Basin to the Northeastern Basin of Florida Bay. To the west of Tabb's Rise lies Taylor Slough while the eastern flow way is comprised of the various smaller sloughs created by flow through the Transverse Glades.

During a typical wet season, water in Taylor Slough flows southwest from Paradise Key toward an area between Cuthbert Lake and Little Madeira Bay (Figure A2.4). However, the Buttonwood Embankment prevents water from flowing freely into these lakes and bays (Craighead 1971). This ridge is particularly well developed along the landward coasts of West, Cuthbert, Henry, and Seven Palm lakes and Madeira and Little Madeira bays (Figure A2.4; Craighead 1971), thereby causing water to pond in a shallow basin north and east of the embankment on the west side of Taylor Slough known as Craighead Pond (Figure A2.4; Craighead 1971, Schomer and Drew 1982). During a typical wet season water enters the northeastern Basin of Florida Bay via several well define creeks in Little Madeira Bay. Further eastward, the Buttonwood embankment is less well developed and fresh water from Taylor Slough flows into western Joe Bay through myriad creeks (Figure A2.4). Massive flow from Taylor Slough would create freshwater conditions within Little Madeira and Joe Bays (McIvor et al. 1994). Freshwater would be passed further south through the mouth of Little Madeira Bay, Dynamite Pass, and Davis, Mud and Trout Creeks (Figure A2.4), thereby diluting the Northeastern Basin (Figure A2.4). Internal circulation and mixing within the basin itself would likely have created brackish water conditions throughout the basin.

During the dry season NESS and the Headwaters Basin dried out and flows southward from Paradise Key and the transverse glades trickled to a stop. The only flow through the coastal wetlands south from the Pinelands (including the Transverse Glades sloughs) would have been caused by local rainfall stemming from infrequent wet cold fronts (Figure A2.5). However, a sizable amount of fresh water remained impounded behind the embankment in Craighead Pond. This freshwater reservoir extended from 1 km to 8 km landward from the embankment (Figure A2.5; Craighead 1971). Water from this reservoir continued to enter the Northeastern Basin via three major creeks (Taylor River, East Creek, and Mud Creek) that penetrate the embankment north and east of Little Madeira Bay (Figure A2.5; Craighead 1971, Schomer and Drew 1982). and through the creeks on the west side of Joe Bay (Figure A2.5; McIvor et al. 1994, Ley et al. 1995). Currently, fresh water continues to flow into the Northeastern Basin well into December or January (Kushlan 1980, pers. obs.), however, prior to 1960, the flow of fresh water is believed to have continued through as late as March (Van Lent et al. 1993, Fennema et al. 1994), if not for the entire dry season (Craighead 1971). These dry season flows coupled with the frequent wind events during the dry season (Holmquist et al. 1989, Barrata and Fennema 1994) helped to maintain a low a salinity environment in northeaster Florida Bay throughout the dry season.

The Northeastern Basin of Florida Bay (Figure A2.6; from here forward I will refer to this basin using the an acronym NEFB for northeastern Florida Bay), is characterized by few mud banks that are relatively poorly developed (Wanless and Tagett 1989) and not as restrictive to internal circulation (Figure A2.6),. However, external circulation with the Atlantic Ocean is obstructed by Key Largo on the east and south. Likewise, circulation with the Gulf is blocked by the well developed bank system to the west. As a result NEFB does not experience diurnal tides (Powell 1987, Holmquist et al. 1989) thereby minimizing marine influences. During the wet season and early dry season, large quantities of fresh water from the Everglades enters NEFB along its northern margin (Schomer and Drew 1982, Baratta and Fennema 1994, McIvor et al.

1994). Wind driven tides can be severe and are the chief mixing agent within NEFB (Holmquist et al. 1989, Wang et al. 1994), having an homogenizing effect on the salinity regime within this shallow basin. The combination of freshwater inflow, internal mixing and reduced communication with external marine influences results in a relatively uniform low salinity throughout NEFB during high runoff periods. For example, ENP has continuously monitored salinity along southern margin of the basin (Middle Butternut Key) since 1993. During high rainfall years, salinity dropped below 20 ppt for prolonged periods with the lowest recording of just under 15 ppt occurring in November 1995.

Model Estimates of Historic Flow through Taylor Slough

Pre-drainage empirical flow data for Taylor Slough does not exist, however, hydrologic models have been developed which are useful tools in estimating historic flows through Taylor Slough. An example of a simple model is presented in Figure A2.7. Water flow has been recorded at the Taylor Slough bridge in ENP since 1960. The bridge is located on State Route 9336 at the point where Taylor Slough leaves the headwaters basin (Figure A2.3). Based on the relationship between these data and water levels recorded at the Homestead Well (located in the headwaters basin east of the ENP boundary; Figure A2.3), Fennema (ENP hydrologists, pers. comm.) back-calculated estimated flow rates to 1932 (Figure A2.7).

Technological advances have made it possible to develop highly sophisticated hydrologic models that dynamically simulate flows throughout the Everglades system (Fennema et al. 1994, Sklar et al. 2002). These models are very useful tools in evaluating historic flow patterns and simulating flows under different water management scenarios. However, these models also have there limitations. One limitation that is significant to this discussion is that models are subject to “edge” effect. This makes them less accurate toward the edges of the modeled area (Fennema pers com, Fennema 1994). Unfortunately, the Taylor Slough drainage area is at the edge of the Everglades system and, as such, usually is subject to “edge” effects. This is not to say that the estimates are useless in Taylor Slough but simply that the confidence limits around the estimated flows are much wider than at more centrally located areas (Shark River Slough for example).

For these reasons, multiple model outputs will be used in estimating historic flows through Taylor Slough. The Fennema model (Figure A2.7) lacks sophistication but it is appealing in that it is not subject to the problem of edge effects. It is also appealing in that the estimated mean annual historic flow is at the low end of the spectrum for Taylor Slough flow estimates. Using only the estimated data from 1932 to 1960 (when the L-29 levee was completed—see below for further justification for using this period), the Fennema Model estimated mean annual flow at 91,000 Acre-Ft per year. At the other end of the spectrum, Van Lent et al. (1993) estimated mean annual historic flows for Taylor Slough to be 162,500. Most recently, Sklar et al. (2002) used the "Natural Systems Model" (currently, the state-of-the-art model that has become the standard used by hydrologists working in the Everglades) to estimate mean annual historic flows at 119,000 Acre-Ft. These three models span the gamut of estimates of pre-drainage Taylor Slough flows and will be used here to evaluate the effect of water management practices on flows to NEFB.

Water management impacts on the Taylor Slough/NEFB watershed

The history of water management of the Everglades began with the efforts of Hamilton Disston in the late 1800's. Since then, a massive canal and drainage system was created that had myriad impacts on the flow of water through the Everglades (Light and Dineen 1994, McIvor et al. 1994, Lorenz 2000, Sklar et al. 2002). The impact of these changes on Florida Bay ranged from minor to severe (McIvor et al. 1994, Lorenz 2000). However, there were two projects that had such major and direct impacts on the flow way to Florida Bay that most of the others pale in comparison.

The first of these was the completion of the L-29 levee adjacent to the Tamiami Trail (US-41) in 1962 (Figure A2.8). The Tamiami Trail itself and the adjacent barrow ditch were completed in 1915 (Sklar et al. 2002), thereby effectively cutting off what is now Everglades National Park off from upstream flows. However, the roadbed was highly porous and the road itself was frequently overtopped during the wet season to the point where it was impassable. In 1961 this transmisiveness ended when reconstruction of the roadbed occurred and the L-29 levee constructed. The effect on flow was immediate and dramatic (Figure A2.7). Prior to the construction of the Tamiami Canal, annual mean flow at Taylor Slough bridge was estimated to be 91,000 Acre-ft per year. Following these events, mean annual flow was measured at 17,500 (Figure A2.7).

The second major effect was the construction and completion of canals in South Dade County beginning in the 1960's and culminating with the completion of the South Dade Conveyance System (SDCS) in the early 1980's (Figure A2.8). Beginning in the early 1960's, canals were dredged through the Transverse Glades (Figure A2.7; C-1, C-102, and C-103) draining them directly to the east coast (Figure A2.8), thereby lowering water levels on both the east and west sides of the Pineland Ridge (Craighead 1971, Simmons and Ogden 1998). Natural flow through the Finger Glades and resulting sheet flow to the eastern side of Florida Bay (eastern Joe Bay to US-1; Figure A2.3) was largely terminated by construction and operation of these canals (Craighead 1971).

Construction of Canal-111 (C-111; Figure A2.7) was completed in 1968 (Johnson and Fennema 1989). This canal was much larger than the others in south Dade as it was also designed to transport rocket engines by barge from the Aerojet plant down the canal to Biscayne Bay and the Atlantic Ocean (Marine 1971). As a result, the capacity of the C-111 canal to move water was greatly increased by constructing it for this purpose (Marine 1971, Carter 1976). The C-111 was also different from the other canals in that it did not follow a naturally occurring channel (e.g. the Transverse Glades) but instead breached the Pineland Ridge at Structure-177 (S-177; Figure A2.7). This breach was very large relative to other canals and presented an unnatural, high capacity means for moving water out of the Taylor Slough drainage basin (Van Lent et al. 1993). The end result was that massive amounts of water could be routed away from former destinations thereby allowing for drainage and flood control in southern Dade County.

The C-111 was further designed to supply flow to the Eastern Panhandle region and eastern Florida Bay, thereby replacing the flow lost by the dredging of the Transverse Glades. This was accomplished by the dredging of gaps in the southern levee of the east-west leg of the canal (Figure A2.8). The plan was that water flow through S-18C would back up behind S-197 (originally just a plug at the southern terminus of the C-111) and be forced out of the canal through these gaps (Ley et al. 1995). Water exiting the gaps would sheet flow southward toward NEFB, thereby offsetting the estuarine water deficit caused by the loss of freshwater flows from the Transverse Glades. Unfortunately, elevation was not taken into account when the gaps were initially installed and the western end of the east-west leg is about 1 ft higher elevation than the east end (Ley et al. 1995). Therefore, almost all water exiting the canal did so through the eastern-most gaps (Light and Dineen 1994, Ley et al. 1995) and the majority of fresh water entering the northeastern basin of Florida Bay arrived at the eastern end of Long Sound (Van Lent et al. 1993, Ley et al. 1995). This fresh water flowed southwestward along the mainline Keys, mixing with coastal waters, thereby providing little or no benefit to the majority of the estuary (Van Lent et al. 1993). In 1965, the L-31N was extended southward and connected to the C-111 at Structure-174 (S-174; Figure A2.8). Structure-173 (S-173), a relatively small flow conduit (a single culvert with a maximum flow capacity of 100 cubic feet per second) was placed as a divide structure between the upper and lower portions of the L-31N near it's former southern terminus.

In 1968, flood control coupled with environmental concerns resulted in the construction of Levee 31W (L-31W) through the headwaters basin just north and east of where Taylor Slough exited the basin (Figure A2.8; Light and Dineen 1994). With the construction of the L-31W managers could use gated control structures (S-174 and S-176) to shunt unwanted water from the East Everglades into Taylor Slough (water entered Taylor Slough from the canal system via 3 culverts at S-175), with the side benefit of supplementing the now limited supply of water reaching the headwaters basin from NESS and the Rocky Glades (Van Lent et al. 1993). The reality was that the canals could not supply enough water to Taylor Slough to offset the loss of flow across the Rocky Glades.

Between 1968 and 1970, ENP expressed concerns about the fresh water needs of Taylor Slough and Florida Bay to the U.S. Congress. The result was legislation that mandated the Minimum Schedule of Water Deliveries (MSWD) plan (Light and Dineen 1994). The MSWD dictated that SRS receive 260,000 acre-ft, Taylor Slough receive 37,000 acre-ft and the C-111 basin receive 18,000 acre ft of water per year (Light and Dineen 1994). The source of this water was to be the Water Conservation Areas (WCA). To execute this order, managers had to make structural changes to the existing canals so that water could be moved from the WCA’s to ENP. These structural changes made up the SDCS (Light and Dineen 1994).

The most substantive component of the SDCS was the addition of the massive (1160 cubic feet per second) Structure-331 (S-331) pumping station, replacing S-173 (Johnson and Fennema 1989, Van Lent et al. 1993, Light and Dineen 1994, Ley et al. 1995). With the completion of S-331 in June of 1983, the regional water supply (represented by inflows to L-31N at S-334 and S-335) could be forced into the lower basin of the L-31/C-111 complex (Johnson and Fennema 1989, Van Lent et al. 1993). The operational design of the SDCS called for the upstream components to shunt the regional supply southward to S-331 (Ley et al. 1995). From S-331, the water was to flow southward on the L-31N to the confluence of the L-31W and C-111. The existing S-174 and S-176 gated spillways (Figure A2.8) were to be operated so that the appropriate amount of water was delivered to the Taylor Slough and C-111 basins according to the MSWD mandate (Van Lent et al. 1993). A second smaller (165 cubic feet per second) pump was located at S-332 (constructed in 1980) on the L-31W and pumped water from S-174 into the Taylor Slough headwaters basin (Van Lent et al. 1993). Ideally, S-18C was designed to pass water from S-176 into the lower reach of the C-111. S-197 was designed to remain closed, thereby forcing water to exit the canal, re-hydrating wetlands north of the canal and sending sheet flow toward the Northeastern Basin of Florida Bay via the gaps in the southern levee (Van Lent et al. 1993, Ley et al. 1995). Aside from water entering NESS from S-333, virtually all water reaching Taylor Slough and northeastern Florida Bay was controlled by state and federal agencies following the completion of the SDCS. The Rocky Glades and the Taylor Slough headwaters basin had been effectively bypassed.

The effects of water management on salinity at the spoonbill foraging grounds

The preponderance of scientific evidence clearly indicates that NEFB has become more saline as a result of water management practices of recent decades (McIvor et al. 1994, Ross et al. 1996, Meeder et al. 1996, Van Lent et al. 1999). It has been postulated that drainage of the Everglades has resulted in decreased flows through Taylor Slough thereby raising the overall salinity in Florida Bay (Van Lent et al 1993, McIvor et al. 1994, Lorenz 2000). This increased salinity has been implicated in many of the adverse ecological changes that occurred to the ecosystem in the 1980’s and early

1990's (Forqueran and Roblee 1999, Lorenz 1999, Mazzotti 1999). Recently, however, the concept that less fresh water is reaching NEFB has been called into question (Brand 2002). More specifically, flow simulations used in the Central Everglades Restoration Plan suggest that there may be more water being delivered to Florida Bay now than there was historically (Restudy Model Runs). The merits of this argument can be easily demonstrated by comparing Figure A2.7 to Figure A2.9. This comparison indicates that in recent years the C-111 canal is delivering more water to NEFB than Taylor Slough did historically. The higher model estimates of 119,000 Acre-Ft (Sklar et al. 2002) and 162,500 Acre-Ft (Van Lent et al. 1993) per year suggest that current C-111 discharges (Figure A2.9) fall within the range of interannual variation of historic flows.

There is little doubt that salinity in NEFB has increased over the last few decades (McIvor et al. 1994, Ross et al. 1996, Meeder et al. 1996, Van Lent et al. 1999). Researchers indicate that this increase in salinity is significantly larger than can be explained by sea level rise (Meeder et al 1996) and that the increase is clearly related to changes in water management practices (Ross et al. 2002). So if the quantity of fresh water reaching NEFB is about the same as the pre-drainage quantity than some parameter other than quantity of flow must have an impact on salinity. The most conspicuous difference is that the distribution of flow has switched radically from Taylor Slough to the C-111/Panhandle region (Figures A2.7, A2.9, and A2.10).

Van Lent et al. (1993) indicated that discharge from the C-111 arrives at Florida Bay primarily to the eastern extreme near US Highway 1 and passes southward as a narrow plume along the Florida Keys. The amount of this flow that mixes with the greater portion of the northeastern basin (as depicted in Figure A2.6) is minimal (Van Lent et al). Therefore, C-111 flow does little to lower the salinity in NEFB. In contrast flow through Taylor Slough reaches Florida Bay further west (through Little Madeira and Joe bays) thereby mixing thoroughly with NEFB and lowering salinity in the basin (Figure A2.6).

I propose that NEFB acts as a salinity buffer for the coastal wetlands. During low freshwater flow periods of the dry season, water from NEFB is blown into the coastal wetlands thereby increasing overall salinity. The degree to which the salinity increases is dependant upon the ambient salinity in NEFB. If the salinity is relatively low in NEFB at the beginning of the dry season, than salinity in the coastal wetlands will remain relatively low throughout the dry season. Based on this concept, freshwater discharge from the C-111 will do little to buffer the salinity of the coastal wetlands.

A comparison of salinity in NEFB to freshwater flow through both Taylor Slough and C-111 can be used to asses the relative impact of each flow way on NEFB salinity. Since 1993, ENP has continuously collected salinity data at two locations in NEFB: Butternut Key and Duck Key (Figure A2.11). Typically, salinity begins to increase in January (see Figure A2.11 in IOP report) so the low salinity in January at each site would indicate the buffering capacity of NEFB. Regression analysis was used to compare antecedent wet season flows from Taylor Slough and C-111 to the January low salinity at Duck and Butternut Keys for each year from 1994 to 2001 (Figure A2.12). These results clearly suggest that flow from Taylor Slough during the wet season has a determinant effect on the salinity within NEFB, while antecedent discharge from the C-111 appears to have no bearing on January salinity. This is particularly striking given that Duck Key is immediately downstream from C-111 but far to the east of Taylor Slough outflows (Figure A2.11).

Similar regressions were made to examine the buffering effect of NEFB on the coastal wetlands. Salinity data from the TR, JB and HC sites within the coastal wetland (Figure A2.11--see main report and Appendix 2 for details) were used to compare maximum dry season salinity within the coastal wetlands to the January minimum at Duck and Butternut keys from 1994 to 2001. The results clearly suggest that the antecedent conditions in NEFB have a determinant effect on how salty the coastal wetlands will get in the dry season (Figure A2.13). Duck Key is in closer proximity to the coastal wetlands and, not surprisingly, had a stronger relationship with the coastal wetlands salinity. However, it is noteworthy that Butternut Key, almost 10 miles south of the mouth of Trout Creek also has a significant relationship with peak salinity in the coastal wetlands (Figure A2.13). This is probably the most clear indication that NEFB acts as a salinity buffer for the Coastal wetlands.

The above discussion leads to the following conclusions. The salinity characteristics of NEFB are determined by flow quantity through Taylor Slough but not from C-111 discharge. During the dry season, NEFB acts as a salinity buffer for the coastal wetlands that serve as the primary foraging grounds for spoonbills. The diversion of flows away from Taylor Slough to the C-111 in recent decades likely explains the observed increased salinity regime in the coastal wetlands even though the total quantity of freshwater reaching NEFB may have changed very little. Based on multiple studies, the altered salinity regime in the coastal wetlands would have a negative effect on the biological productivity of the wetlands. This effect culminates in a depauperate prey base for higher trophic levels (using Roseate Spoonbills as a representative example) that forage in this wetland system and depend on its resource for reproductive success. The net result was an overall decline in populations of predators within the region. A consequence of these conclusions is that it calls into question a widely accepted concept (one previously championed by the author) that in order to restore the historically low salinity regime of NEFB, more freshwater must be delivered to NEFB, regardless of source.

The effects of management on water levels at the spoonbill foraging grounds

The effect that the completion of the SDCS had on discharge from the gaps in the C-111 is readily apparent (Figure A2.9). Prior to 1981 the average flow exiting C-111 through the gaps was about 42,000 Acre-feet. Following operation of the SDCS and associated events (1981), the average discharge more than tripled to 140,000 acre feet (Figure A2.9). These differences are extreme and would have widely different effects on the immediate downstream hydrology (Bjork and Powell 1994, T. Smith pers. com.) and flows to the Panhandle region (Barrata and Fennema, 1994).

The major question that needs answered is how do the pre- and post-SDCS discharges from the C-111 compare to the historic flows deliveries from the Transverse Glades? In the absence of empirical data, simulated flows from hydrologic models must be used. However, this raises more problems in that multiple models have been developed and few agree with one another on the historic flow rates through Taylor Slough (much less the panhandle). The reason for the disparity in model outputs is likely due to model edge effects. Taylor Slough and the Panhandle are at the edge of the Everglades system and, as such, are much more hard to predict than the central regions (Shark River Slough for example). Estimated Taylor Slough flows range from 91,000

Acre-Ft per year (Fennema pers comm, Figure A2.7) to 162,500 Acre-Ft per year (Van Lent et al. 1993). Most recently Sklar et al (2002) published an estimate that is somewhat median to these two extremes (119 Acre-Ft per year). The three model estimates of Taylor Slough flow (91,000, 119,000 and 162,500 Acre-Ft per year) were used to calculate a range of possible historical flows to the Panhandle Region via the Transverse Glades. The historic ratio of Taylor Slough to eastern panhandle flow is not well understood but several estimates of this ratio can be found in the literature. The lowest estimated ratio of Taylor Slough to Panhandle flows was that mandated by U.S. Congress under the MSWD (2:1 ratio). Van Lent et al. (1993) estimated a 7:3 ratio while Fennema et al. (1994) estimated a 3:1 ratio. The largest published ratio was that of Sklar with a 4:1 ratio of Taylor Slough to Panhandle flows. Table A2.1 summarizes these estimates of Taylor Slough flows and panhandle ratios. By product of the estimated flows with the ratios provides a range of estimated flows to the panhandle (Table A2.1). Using this simple method, the minimum estimate of historic flows through the Transverse Glades to the Panhandle was 18,200 Acre-ft and the maximum was 53,000 Acre-ft. The mode for these estimates fell between 30,000 and 32,500 Acre-ft (Table A2.1). Flow through the C-111 prior to SDCS was 42,000 Acre-ft which falls well within the range of these estimates (Figure A2.9). In contrast annual flow following the SDCS (140,000 Acre-ft) is an order of magnitude higher than the smallest of these estimates and almost three times the largest (Figure A2.9). Based on these figures, it is clear that recent (and current) water management practices result in C-111 discharges far in excess of the natural flow from the Transverse Glades. Furthermore, the C-111 discharges prior to the SDCS may have been indicative (or at least close to) the historic flows through the Transverse Glades.

Dry season discharges from the C-111 are of particular concern because these impact water levels on the coastal wetlands and result in reversals in dry down patterns (Barratta and Fennema 1994, Bjork and Powell 1994, Lorenz 2000). Several studies have calculated wet to dry season ratios based on rainfall patterns. For example, Van Lent et al. (1993) estimated that dry season flow was 15% of wet season flow based on rainfall patterns in Everglades National Park. This seems a logical progression, however, rainfall estimates vary from location to location and there is no good justification for selecting one location (or region) over another. Using official NOAA rainfall data from 25 locations in the their Everglades and Lower East Coast Region, dry season rainfall (December through May) was about 30% of the annual total. Duever et al. (1994) estimated rainfall for the entire southern peninsula and concluded that the dry season accounted for 25% of the annual total. Using these three estimates (15%, 25% and 30%), historic dry season flow for the panhandle was estimated using the minimum, maximum and modes of total annual flow from Table A2.1. Results indicate that annual dry season flows were between 2730 and 16,088 Acre-Ft, with mode estimates of 7500 and 8044. Actual mean dry season discharge form C-111 prior to the completion of the SDCS was 6850, which falls comfortably within the estimated historic flows. Following the SDCS, discharges swelled to 31,500, more than twice that of the largest estimate and more than 10 times that of the smallest. This change in dry season flow is presented graphically in Figure A2.16. It is distressing to note that dry season flows after completion of the SDCS exceeded wet season flows prior to the SDCS.

The impact of increased dry season discharges from the C-111 on spoonbill nesting activity is evident. Spoonbill nesting success decreased more than 50% following the SDCS. The number of active colonies in NEFB decreased from 7 in 1978 to 2 in 2002 and the number of nests declined from almost 700 in 1978-79 to less than 100 in 2001-02. The effect of C-111 discharge on success can also be demonstrated. Spoonbill nesting attempts since 1990 were separated into failed years (<1 chick per nest fledged) and successful years and the total C-111 discharge for the critical 42d post-hatch period was summed for each attempt (Figure A2.17). These results clearly indicate that dry season pulse discharges from the C-111 result in spoonbill nesting failure.

This discussion of the effects of C-111 discharges on dry season water levels in the coastal wetlands leads to the following conclusions. Prior to water management practices, the coastal wetlands were short hydroperiod wetlands. Roseate Spoonbills successfully nested in Florida Bay by using this wetland as a primary foraging ground. This utilization indicates that prey were predictably available during the nesting season. Following management the hydropatterns in the wetland were effected by discharges from the C-111 canal. Out-of-season pulse releases from the C-111 canal result in rising water levels on the wetland (Barratta and Fennema 1994, Bjork and Powell 1994, Lorenz 2000.). These reversals in the dry down process allows prey base fish to escape across the wetland, i.e., the prey becomes unavailable. Spoonbills nesting fails when such releases are significant enough to cause increases in water level. Following completion of the SDCS, discharges through the C-111 increased dramatically causing frequent nesting failure and colony abandonment in NEFB. These discharges continue to present with 6 of 10 spoonbill nesting attempts resulting in failure. Curtailing out of season releases would return the wetlands system to a more natural hydroperiod and promote the return of top predators dependent on the dry down cycle. A consequence of these conclusions is that it calls into question a widely accepted concept (one previously championed by the author) that freshwater from upstream parts of the system should continue to flow through the Panhandle region well into the dry season (at least through February).

Conclusion

The evidence presented indicates that, prior to Everglades water management practices, the coastal wetlands were characterized by lower salinity conditions through the dry season when compared to modern conditions and water levels remained relatively lower throughout the dry season as well. The success of spoonbills in the natural system and their failure in the managed system indicates that these criteria are critical to a normally functioning ecosystem and that restoration efforts should attempt to replicate these conditions. Since spoonbills are sensitive to these conditions and their performance readily monitored and quantified, they should be used as a primary indicator for the success of restoration.

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Model	Estimate TS Flow Total Annual Ac/Ft	Estimated historic Ratio TS to Pan Handle: Min Flows 2.0:1	Van Lent et al. 1993 2.3:1	Fennema et al 1994 (NSM) 3.0:1	Sklar et al 2002 (NSM) 4.0:1
Van Lent et al. 1993	162500	**53625**	48750	40625	**32500**
Sklar et al 2002	119000	39270	35700	29750	25800
Fennema (unpub)	91000	**30000**	27300	22750	**18200**

Table 1. Estimated mean annual historic flow to the panhandle based on model outputs for Taylor Slough. Bold type indicates minimum, modes and maximum estimates. Actual mean annual discharge from C-111 measured prior to the completion of the South Dade Conveyance System was 41,986 ($\pm$ 11,174) Acre-Ft which falls within the minimum and maximum estimates from the models. In contrast, post SDCS C-111 mean annual discharge was 140,191 ($\pm$ 10.840) Acre-Ft, more than twice that of the estimated maximum. These estimates indicate that the SDCS has resulted in excess flows to the panhandle region.

Estimate	Estimate PH Flow Total Annual Ac/Ft	Estimated flow during dry season Van Lent et al. 1993 15%	Duever et al 1994 25%	NOAA Rainfall Data 30%
Max	53625	**8044**	13406	**16088**
Mode	32500	4875	8125	9750
Mode	30000	4500	**7500**	9000
Min	18200	**2730**	4550	5460

Table 2. Estimated mean dry season historic flow to the panhandle based on Minimum, Modes and Maximum estimates from Table 1. Bold type indicates minimum, modes and maximum estimates for dry season. Actual mean annual discharge from C-111 measured prior to the completion of the South Dade Conveyance System was 6850 (± 2224) Acre-Ft which falls within the minimum and maximum estimates from the models. In contrast, post SDCS C-111 mean annual discharge was 31,497 (± 7096) Acre-Ft, almost twice that of the estimated maximum. These estimates indicate that the SDCS has resulted in excess flows to the panhandle region during the dry season nesting period for wading birds.

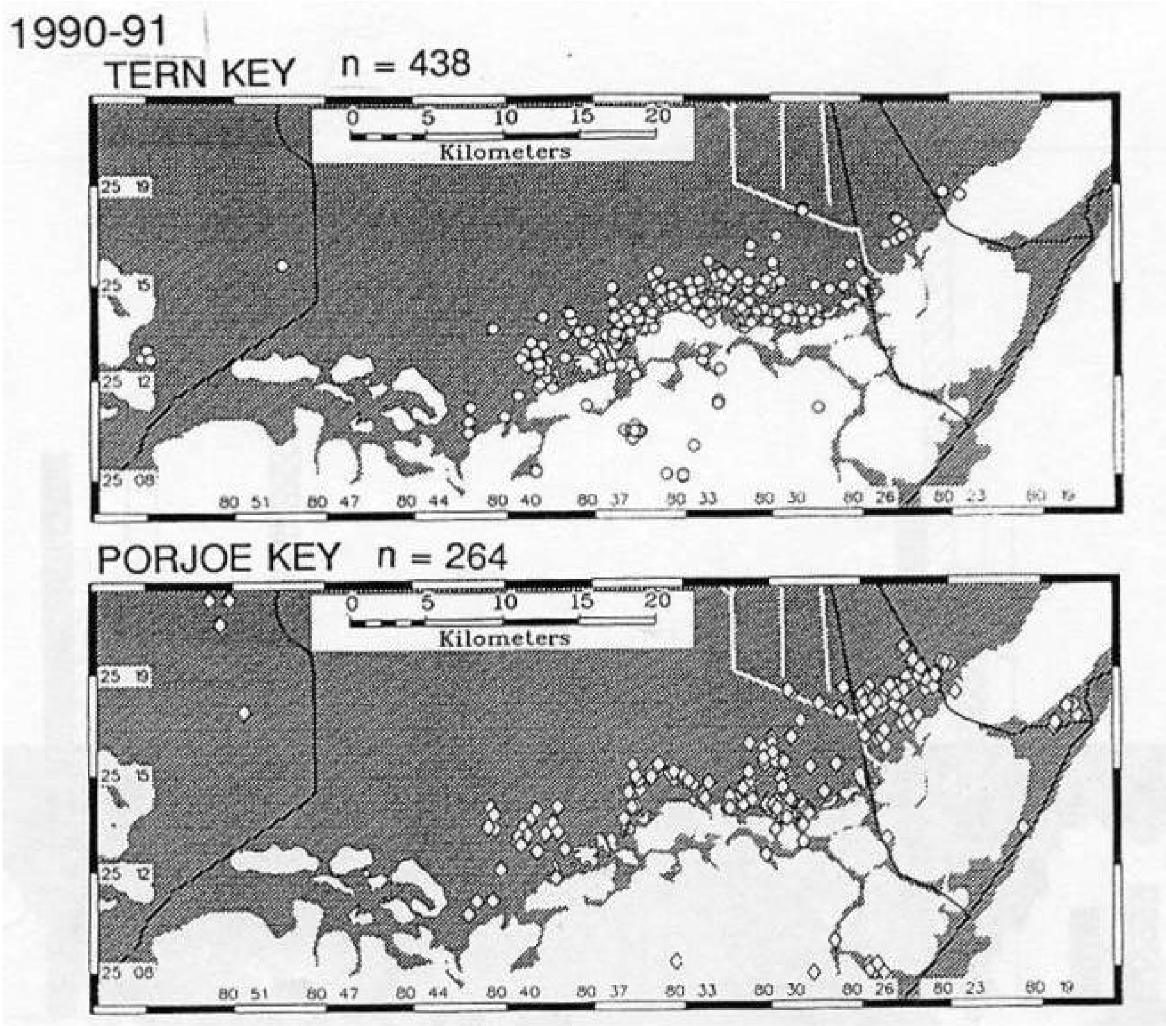

Figure A2.1. Foraging location of spoonbills in 1990-1991. Individual birds were followed from their nesting colonies to there foraging location using a fixed wing aircraft (Bjork and Powell 1994). These data indicate that the primary foraging area for these birds was the coastal wetland from Taylor Slough to Barnes Sound. Note the heaviest concentrations are effected by C-111 discharges.
Figure A2.2. Historical geologic and hydrologic features of the Northeastern Shark River Slough to Northeastern Florida Bay flow way.

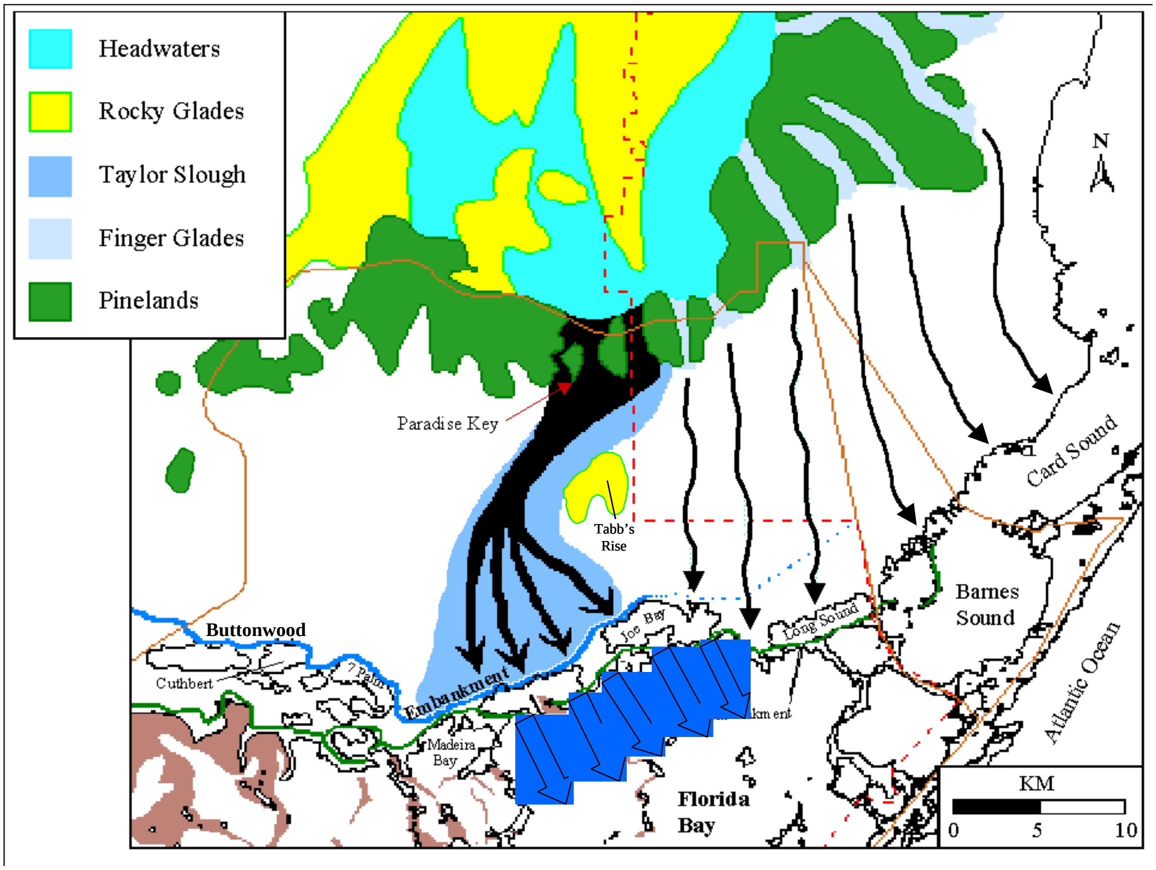

Figure A2.3. Stylized distribution of freshwater flow to NEFB during the wet season.

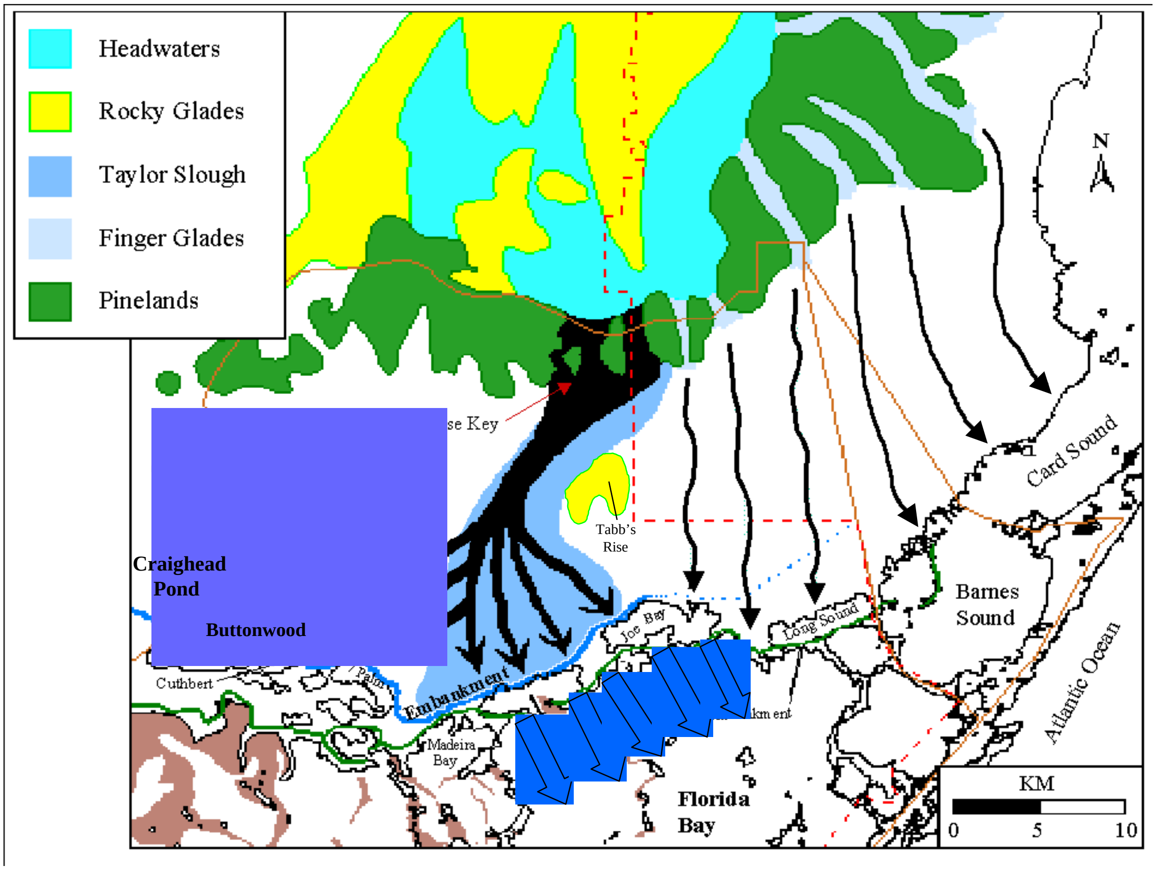

Figure A2.4. Taylor Slough wet season flow. Historically flow through TS ponded behind the Buttonwood Embankment and recharged Craighead Pond. The vast majority of water entering Florida Bay during the wet season di so through Little Madeira Bay and Joe Bay. The lack of banks in the Northeastern Basin promoted mixing resulting in brackish conditions through out.

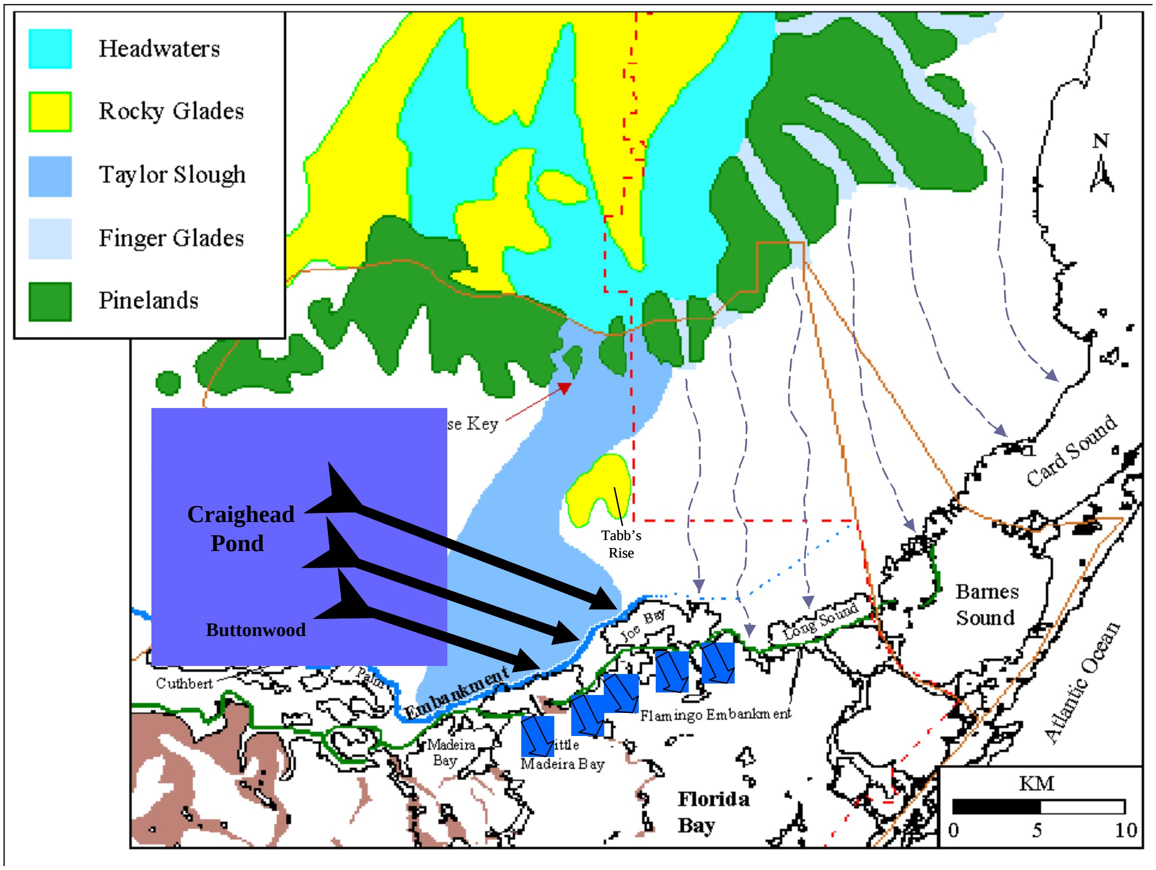

Figure A2.5. Dry season flow. With the sesation of fall rains, water levels in the Headwaters Basin subsided and overland flow through Taylor Slough and the Transerse Glades ceased. However, freshwater stored in Craighead Pond continued to flow into L.Madeira Bay and Joe Bay well into the dry season, thereby keeping salinity in NEFB low. Overland flow in the upperTaylor Slough and the Transverse Glades was restricted to local rainfall events which occur infrequently during the dry season.

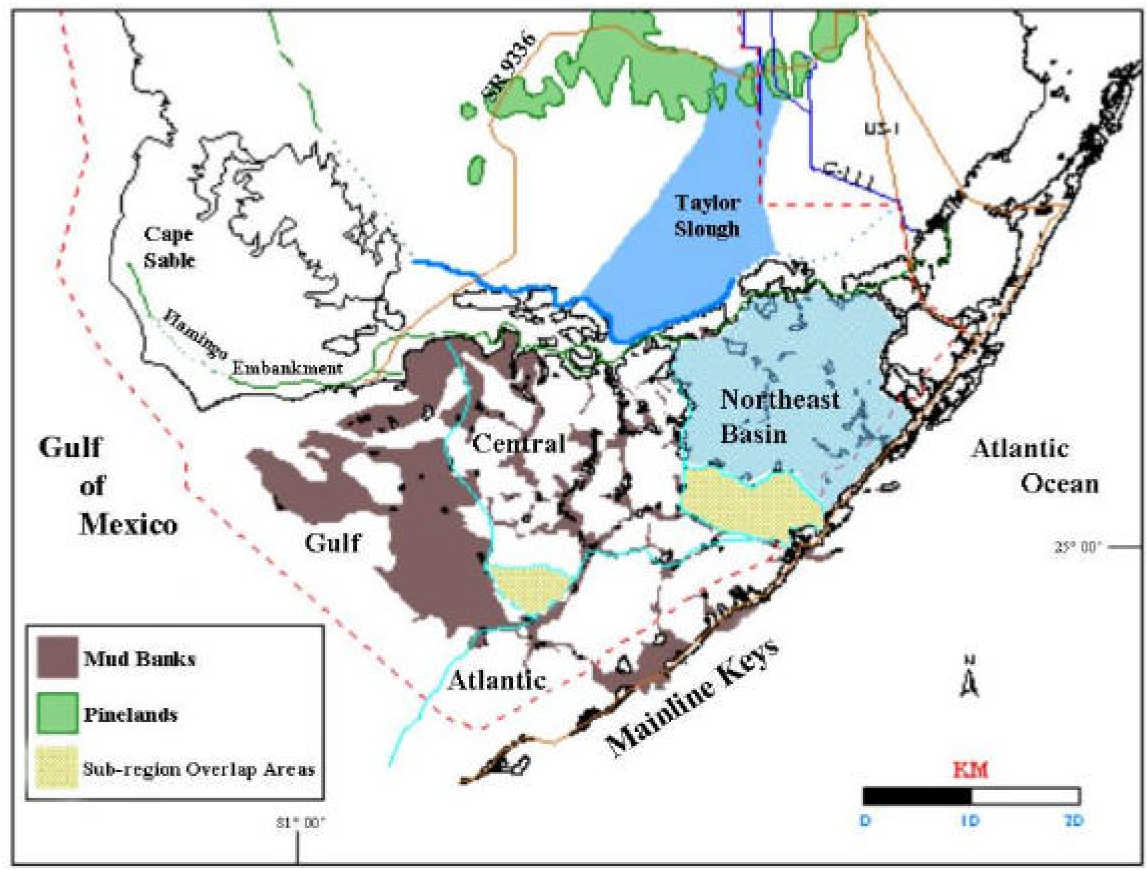

Figure A2.6. During the wet season, the Northeastern Basin acts as a catchment basin for runoff from the Taylor Slough and Panhandle regions. The resulting lowered salinity acts as a salinity buffer for the coastal wetlands during the dry season.

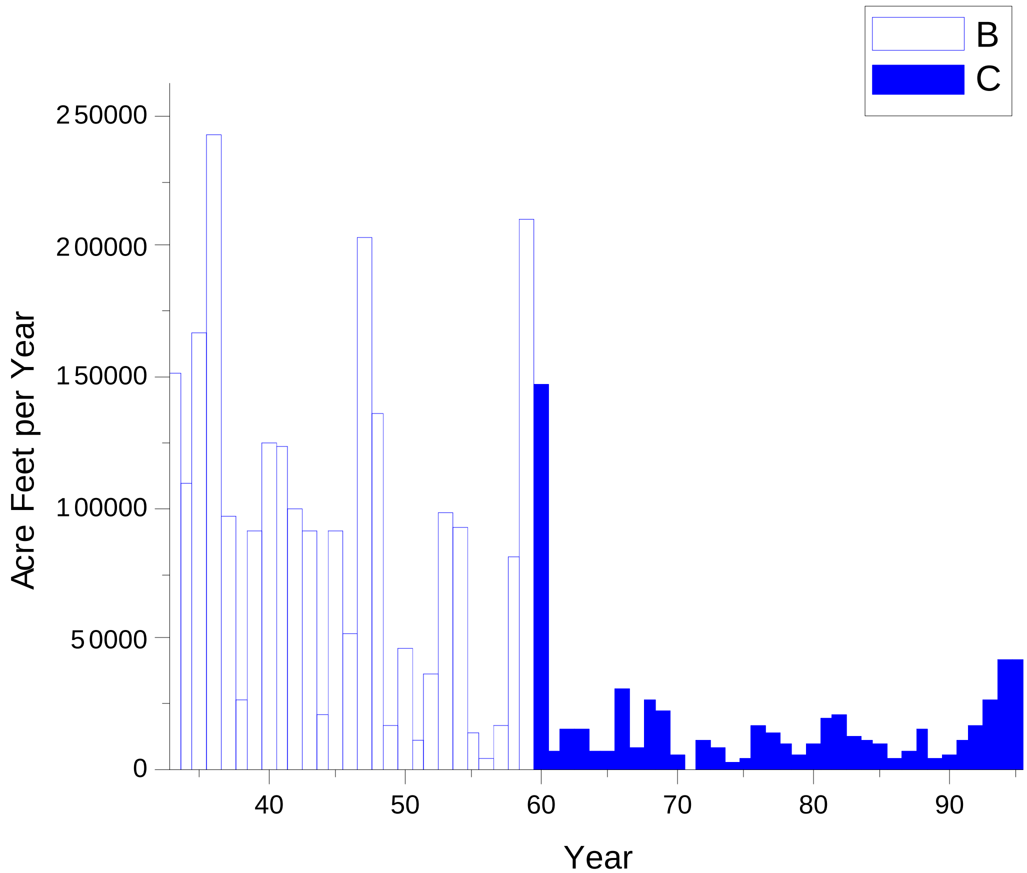

Figure A2.7. Flow through Taylor Slough at the SR 9336 Bridge in Everglades National Park. Data from R. Fennema, ENP hydrologist. Solid bars are data collected from 1960-1995. Open bars are an estimate of flow rates from 1933-1960 based on regression analyses between water levels at Homestead Well and the data collected from 1960-1995.

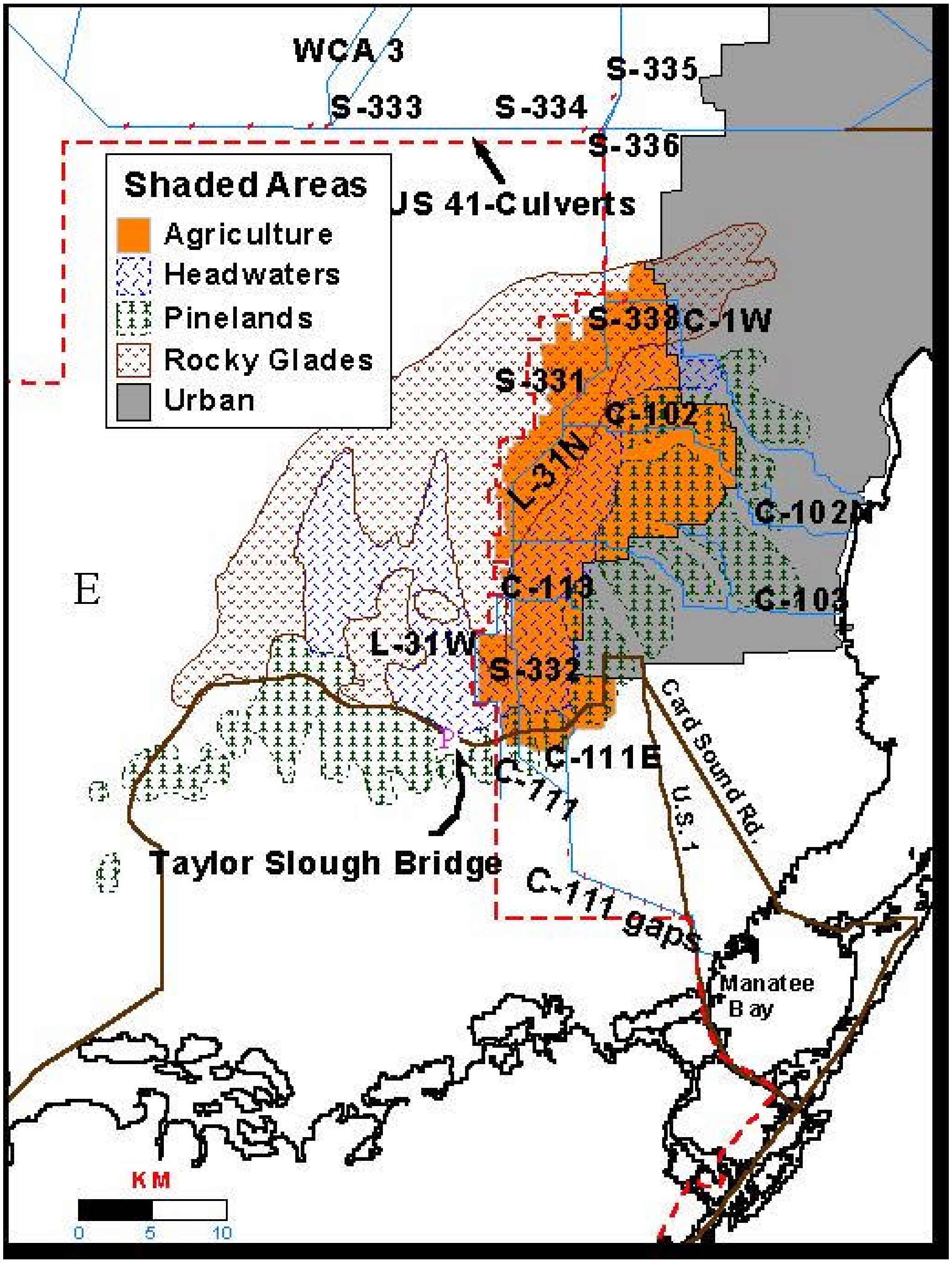

Figure A2.8 Canals and structures directly affecting the Taylor Slough drainage area as planned during the C&SF Project.

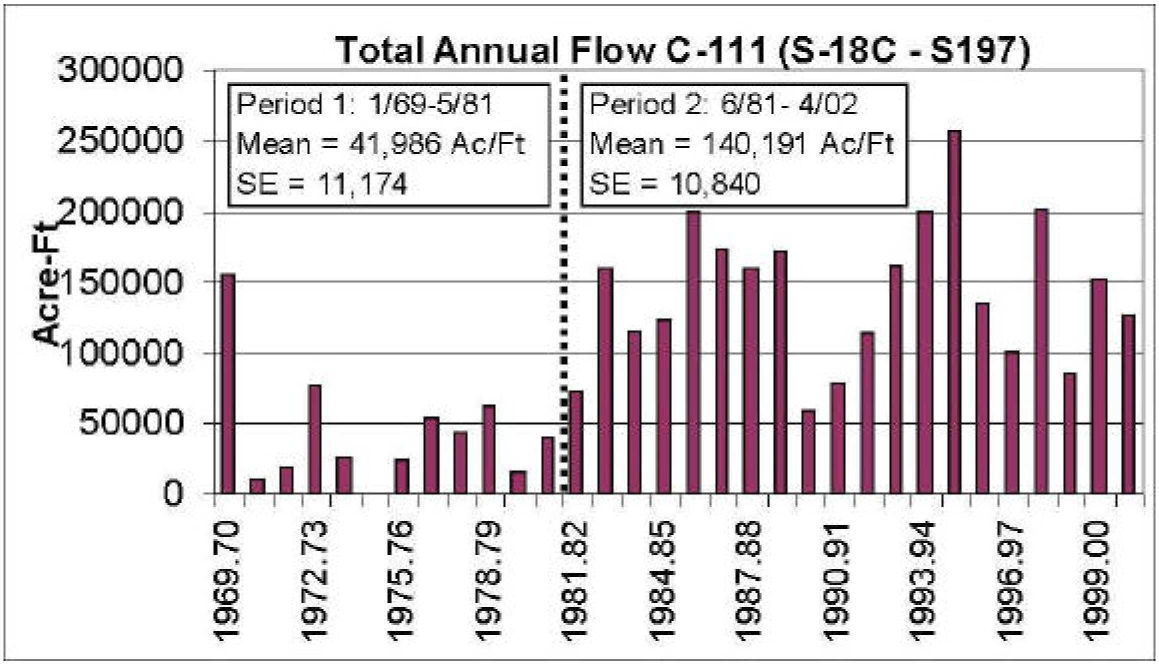

Figure A2.9. C-111 discharge from 1969 through 2001. The difference between Period 1 and Period 2 is the completion of the SDCS and associated activities.
Note that the SDCS was not fully operational until 1984 but between 1981 and 1984 major changes were made in water delivery patterns in C-111 (see below). These three years were analyzed separately from the 1969-1980 period and 1981-2002 period. The three year period was significantly different from the early period but not from the latter period and therefore was included with the latter period for clarity of presentation.

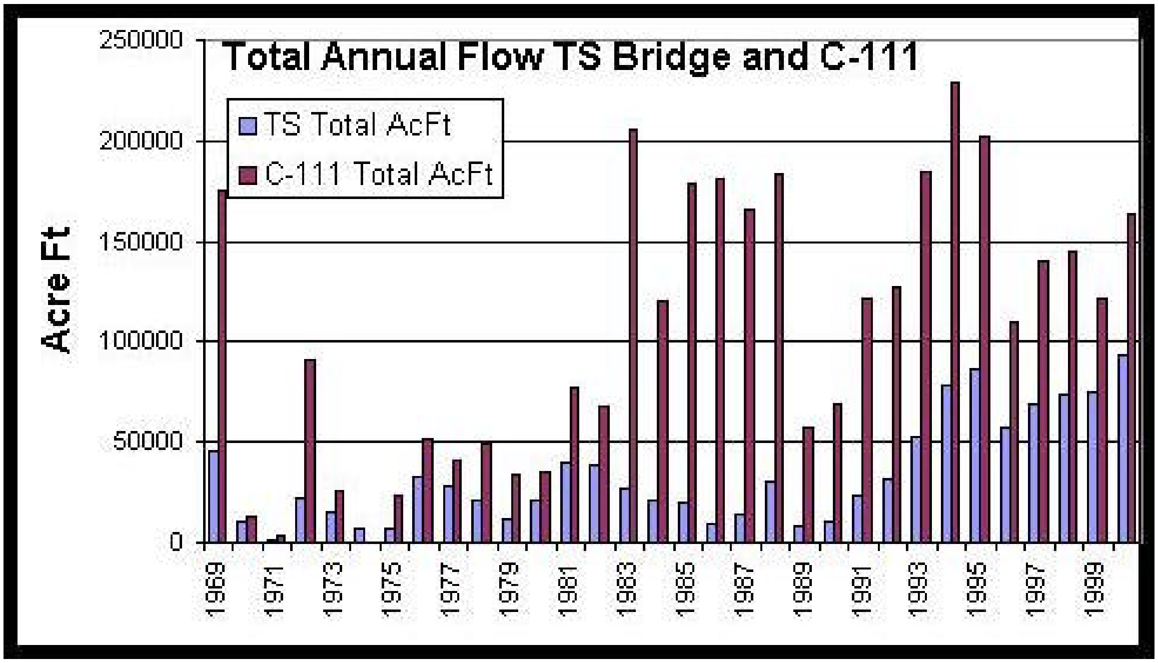

Figure A2.10. C-111 flow = S-18C flow-S-197 flow.A comparison of annual flows through Taylor Slough and Discharge from the C-111 toward Florida Bay

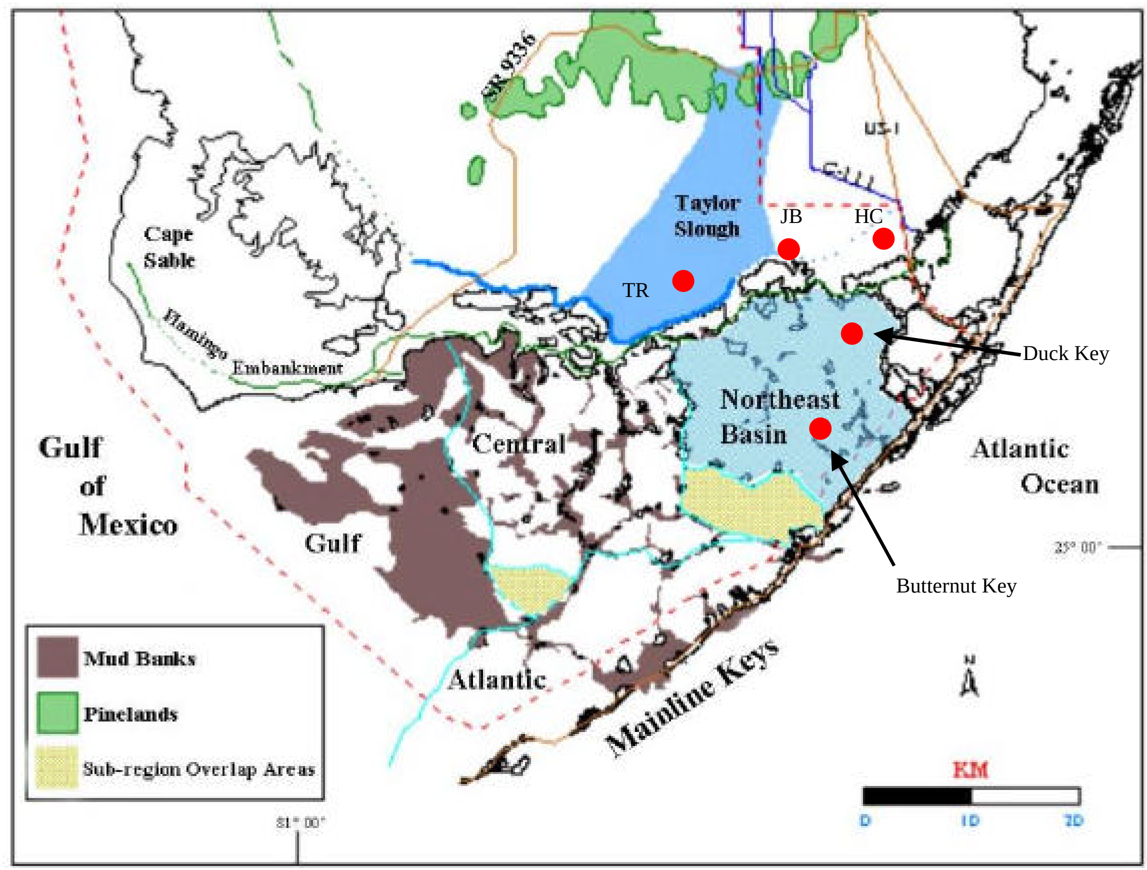

Figure A2.11. Salinity monitoring stations in NEFB and adjacent coastal wetlands.

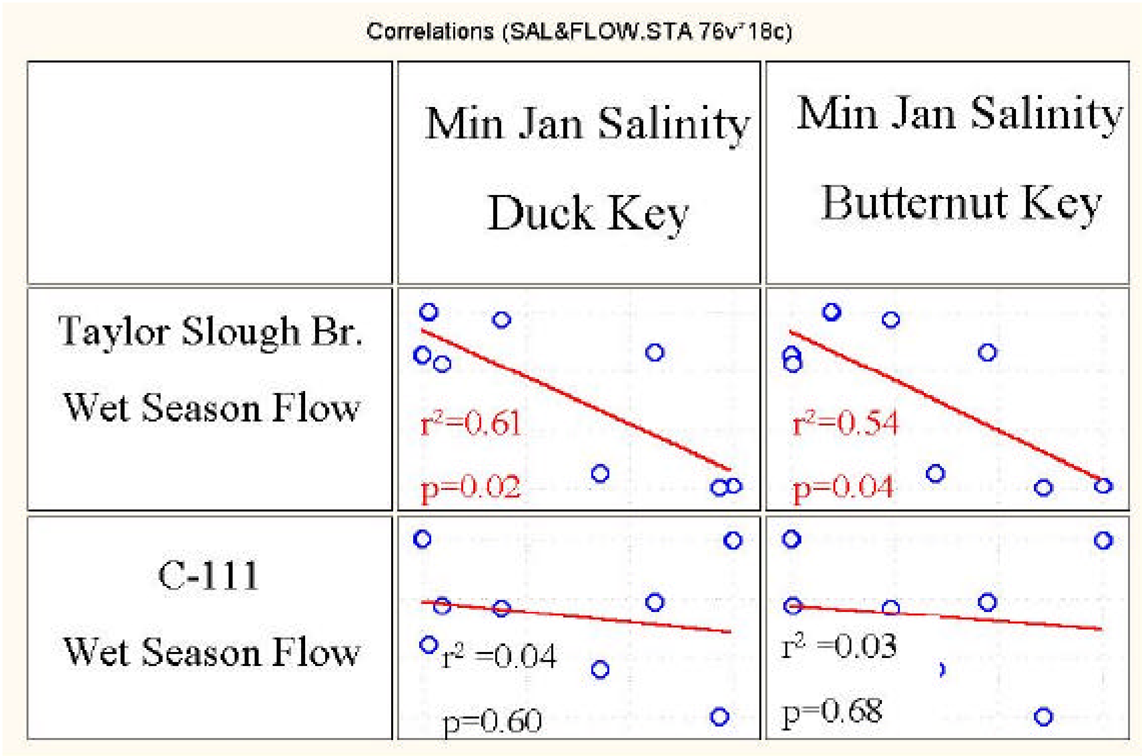

Figure A2.13. Comparisons of salinity at Butternut and Duck keys with Taylor Slough and C-111 flow. Salinity at these two locations is typically lowest in the month of January as there is a delay between the cessation of wet season run-off and an increase in evapotranspiration causing salinity to increase. As can be seen from this analysis, it is runoff from Taylor Slough that is correlated with low salinity in the NE Basin and not C-111 discharge.

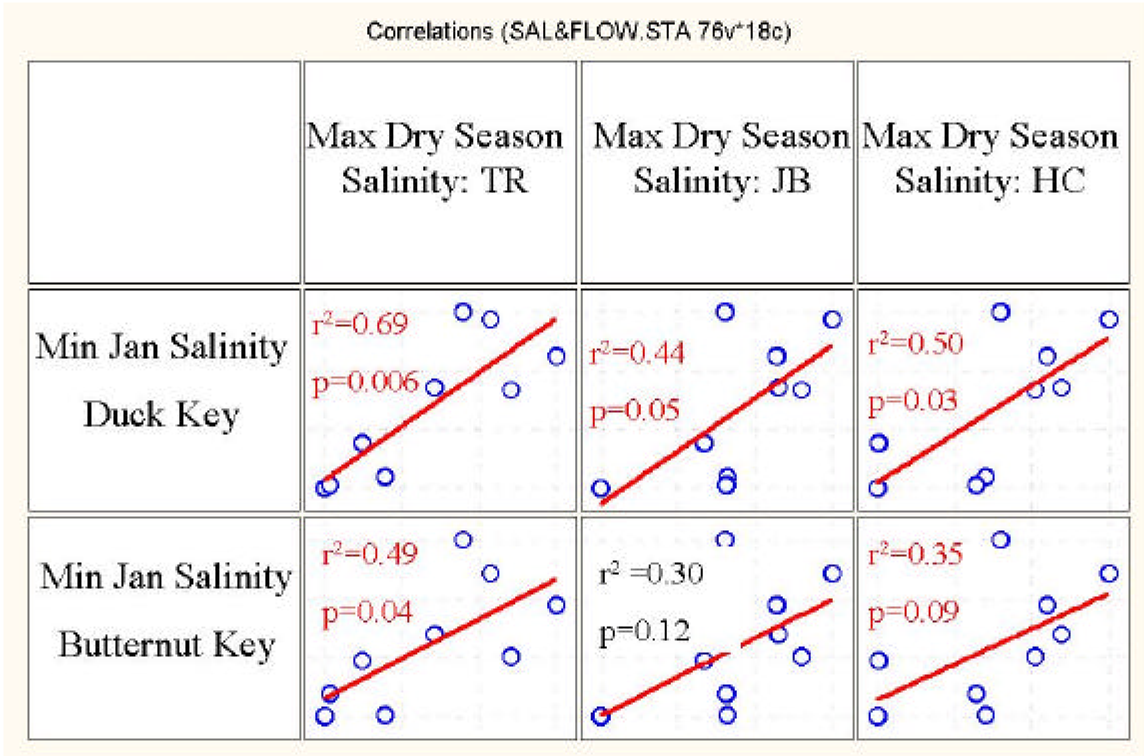

Figure A2.14. The salinity buffering capacity of NEFB as indicated by the relationship between bay salinity and wetland salinity.

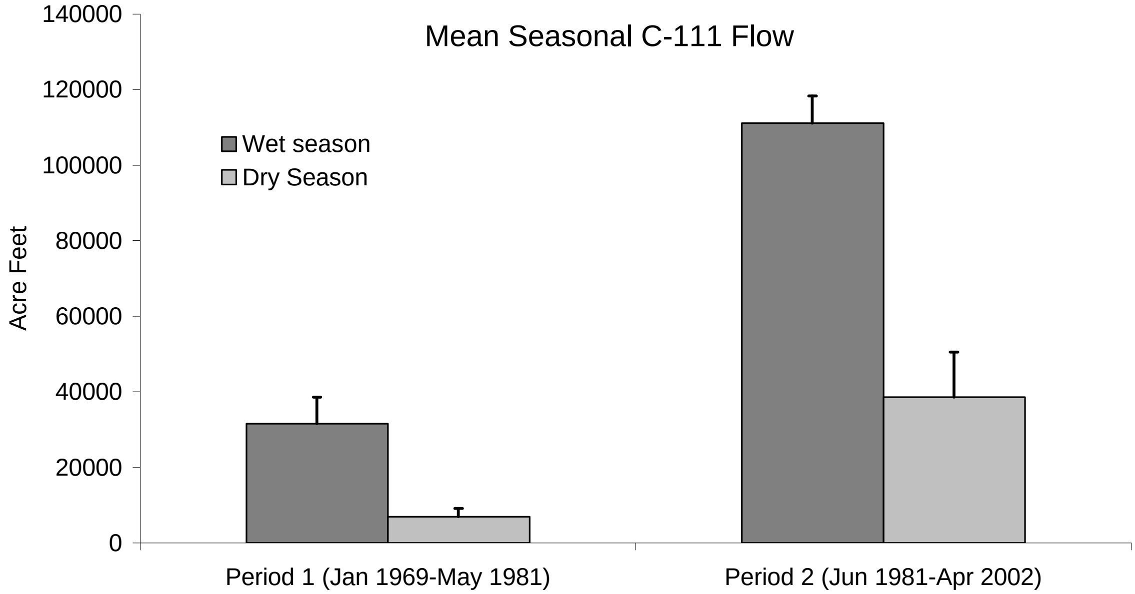

Figure A2.15. Seasonal discharges from the C-111 canal before and after the completion of the SDCS.

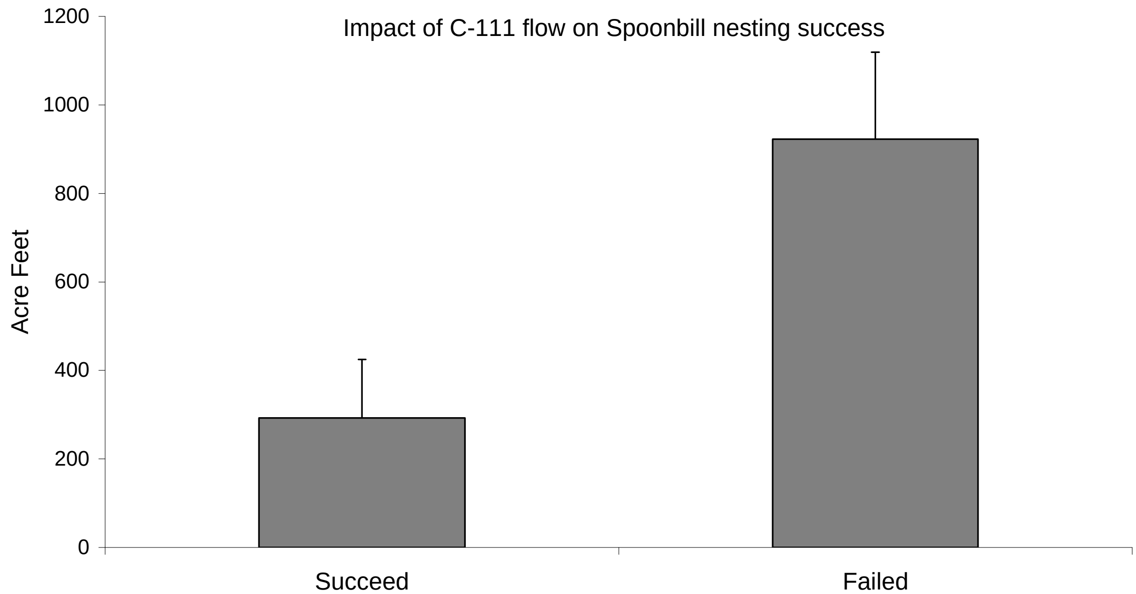

Figure A2.16. The effect of C-111 discharges on spoonbill nesting success.

APPENDIX 3: DATA COLLECTION METHODS

Hydrology and Hydrography. Beginning in November 2000, hydrostations created by Remote Data Inc. were established at the sampling sites. These hydrostations use Hydrolab sensors to continuously monitor water level and salinity. Daily averages were used directly in statistical analyses. Any missing data was supplemented with ancillary data as described in subsequent paragraphs.

Prior to November 2000, water level and salinity data were collected as follows. Water levels were continuously monitored at each site using a Telog brand 2108 potentiometric recorder with a float and pulley design. Accuracy of the equipment was checked by comparing the current measurement of the recorder against a nearby staff gauge each time the system was downloaded. Gaps in the water record were supplemented with regression adjusted data from the nearby ENP hydrostation. Regression models between the Telog data and ENP hydrostations at TR, JB, HC and BS explained 98%, 88%, 86%, and 95% of the variability, respectively. All regression equations were significant ($p<0.01$) and a review of residuals indicated that none of the assumptions for regression analysis were violated. Water level was analyzed using a 2-way ANOVA to determine the difference between 2001-02 and 1999-00 for both months and sites.

Salinity was measured at the site on the day of fish collections using an optical refractometer. These data were strongly correlated to the data recorded by the nearby ENP hydrostations, however there were differences between the two methods. Therefore, the continuous records from the ENP hydrostations were refined to better reflect conditions at the sites through the application of regression models. The hydrostation at TR was located immediately adjacent to the site and was found to be strongly correlated with the refractometer data ($r^2=0.96$). Breakpoint regression was used to convert hydrostation data at JB and HC because the relationship between the refractometer and hydrostation changed between the wet and dry seasons. The breakpoint and regression equations were derived using the computer program 'Statistica' (Statsoft 1995). The conversion models generated by this method explained

92% and 84% of the variability between the refractometer and hydrostation data sets for HC and JB respectively. All regression equations were significant ($p<0.01$).

Salinity at BS was found to fluctuate diurnally with tidal exchange. As a result, the correlation between the single refractometer reading collected on the day of fish collection and the average daily salinity generated by the hydrostation were not as clear as at the other sites ($r^2=0.54$). However, based on close spatial proximity, conditions recorded at the hydrostation were assumed to accurately reflect conditions at the site. The strong relationship between mean daily water level recorded at the station and the site ($r^2=.95$) lends strong support to this assumption. Therefore, the salinity regression was used as a predictor of on site salinity. Although the low precision of this model may result in slight miscalculations in the estimation of on site salinity on a daily level, the impact this has on the long term trends are likely to be insignificant.

Fish Collections. A $9m^2$ drop net method was developed for this study and has been demonstrated to be an effective and unbiased sampling method (Lorenz et al. 1997). Nets were set up, left over night and deployed the following day within 2 hrs after sunrise. Each net surrounded an individual dwarf mangrove tree, thereby sampling both prop root habitat and the open area between trees. Fish were cleared from the net using the fish toxicant rotenone. After about 24 hrs, any fish missed in the initial collection were found floating on the surface and added to the sample. Net clearing efficiencies ranged from 78% to 90% for the most common fish species and averaged 86% for all marked and recaptured fish (Lorenz et al. 1997). All fish collected were taxonomically identified, weighed and measured. Weights for specimens collected during the second day collection were calculated from length-weight regressions generated from first day collection fishes.

Larger transient fishes bias estimates of resident fish biomass toward relatively higher numbers. In order to focus on only resident fish community, larger transient fishes had to be removed from the data set. Resident fishes were defined as those that utilized the entire dwarf mangrove habitat. The largest fishes in collections from the flats nets were about 6.5 cm. Fish larger than 6.5 cm were regularly collected in the creek nets indicating that fish larger than 6.5 cm did not reside in the flats microhabitat.

Using this criterion, all fish larger than 6.5 cm were removed from the data prior to statistical analyses.

Three nets were collected in each microhabitat (creek and flats) at each site, for a total of six nets per sample. The relatively small variance within microhabitat versus the substantial variance between microhabitat (i.e., creeks generally had more fish than flats) indicated that this type of stratification was necessary (Snedacor and Cochran 1967). Sample collections were made in June, September, and monthly from November through April for both 2001-02 and 1999-00.

A stratified mean biomass and number of fish per m^2 for each site were calculated using the method of Cochran (1963). Calculation of a stratified estimate required that the weight of each strata (in this case the wetted area of the creeks and the flats respectively) be known. A complicating factor was that the wetted areas for each of these strata changed through time due to seasonal and climatic water level fluctuations. In order to estimate wetted area of the strata at any given water depth, remote sensing techniques were employed. High altitude false color images (1:1800 scale) depicting 1 km^2 of wetland centered on each sampling site were acquired from ENP. Using the Idrisi GIS package (Eastman, 1995), each image was separated into several discrete color bands representing small incremental changes (on the order of 5 cm) in wetland surface elevation. Each site was then physically surveyed to determine the depth range of each color relative to the continuous water level recorder. When the GIS data were combined with the water level record, the wetted area for each strata could be estimated and used to calculate the weighted mean for each sample collected. Therefore each 6 net sample resulted in a single estimate for biomass and density thus avoiding pseudoreplication (Heffner et al. 1996).

Roseate Spoonbills. Thirty-two of Florida Bay's keys have been used by Roseate Spoonbill's as nesting colonies (Lorenz et al. 2002). These colonies have been divided into five distinct nesting sub-regions based on each colony's primary foraging location (Lorenz et al. 2002). Colonies in the Northeastern (NE) sub-region feed in coastal wetlands described in the previous sections (Bjork and Powell 1994). Consequently, these colonies can be considered the 'experimental' or 'impacted' group when evaluating the effects of water management practices. Following completion of the SDCS, spoonbill nesting declined in the NE sub-region and increased in the Northwestern (NW) sub-region. Birds from the NW sub-region forage in the wetlands on Cape Sable (Bjork and Powell 1994); an area largely free from direct influence of water management practices. For our purposes, the NW sub-region acts as the 'control' group when examining water management impacts.

Prey availability is defined as the mean number of fish collected in the strata with the highest abundance (as opposed to the stratified means presented above which estimate a per m^2 mean across the entire potential wetted area). In other words, prey availability is a measure of the prey concentration effect caused by the drawing down of wetland surface water. Prey availability data was estimated for the NE colonies from fish collections made at the four northeastern coastal sites (TR, JB, HC, and BS; Figures 4-7). In addition, fish samples were also collected at a site on Cape Sable adjacent to Bear Lake (BL; Figure 1) using identical methodology to the northeastern sites. This location is a popular feeding area for spoonbills nesting in the NW sub-region (Bjork and Powell 1994), so BL fish and hydrologic data will be used to evaluate prey availability for these colonies.

Complete nest counts were performed at Tern Key and Sandy Key, representing the NE and NW sub-regions, respectively. Nest counts were performed by entering the active colony and thoroughly searching for nests. Nesting success was estimated through mark and re-visit surveys. These surveys entail marking approximately 50 nests shortly after full clutches had been laid and re-visiting the nests on an approximate 2 week cycle to monitor chick development.